Australian Government

APPROPRIATENESS OF A 350 MILLION LITRE BIOFUELS TARGET

REPORT TO THE AUSTRALIAN GOVERNMENT
DEPARTMENT OF INDUSTRY, TOURISM AND RESOURCES

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Preface

In July 2003 the Commonwealth Scientific and Industrial Research Organisation (CSIRO), jointly with the Bureau of Transport and Regional Economics (BTRE) and the Australian Bureau of Agricultural and Resource Economics (ABARE) were asked to investigate the appropriateness of maintaining an objective that biofuels, produced in Australia from renewable resources, contribute at least 350 ML to the total fuel supply by 2010. The joint study was initiated on 5 August 2003 with a final report required to be delivered on 19 December 2003.

“Appropriateness” was to be considered in terms of net environmental benefits, net economic benefits, net regional benefits and industry viability. In turn, industry viability was to take into account announced reforms to fuel tax arrangements, including the phase out of effective excise relief. As part of the 2003–04 Budget, the Australian Government announced reforms of current fuel tax arrangements to bring all currently untaxed fuels, including ethanol and biodiesel, into the excise and duty system from 1 July 2008. The aim of the announced reforms was to establish a broad sustainable taxation framework for fuels, by addressing anomalies in the fuel tax system and to provide increased long term certainty for investors.

On 16 December 2003 the final excise rates to be applied to ethanol and biodiesel from 1 July 2008 were announced. These rates, to be effective from 1 July 2012, are 19.1c/L for biodiesel and 12.5c/L for ethanol. The conclusions reached in this analysis regarding the economic viability of ethanol and biodiesel production were based on the phasing out of effective excise relief, and hence the conclusions should be interpreted in the light of this.

19 December 2003

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ACRONYMS

ABARE	Australian Bureau of Agricultural and Resource Economics
ABS	Australian Bureau of Statistics
ACCC	Australian Competition and Consumer Commission
ACG	Apelbaum Consulting Group
ADO	Automotive Diesel Oil
ADR	Australian Design Rule
AFR	Air-fuel ratio
AGA	Australian Gas Association
AGO	Australian Greenhouse Office
AIP	Australian Institute of Petroleum
ANL	Argonne National Laboratory
API	American Petroleum Institute
AUDC	Australian Urban Drive Cycle
AUSTEM	Australian Trade and Environment Model (ABARE)
bbl	Barrel
BD	Biodiesel
BD5	Fuel mixture of 5% biodiesel and 95% conventional diesel
BD20	Fuel mixture of 20% biodiesel and 80% conventional diesel
BTRE	Bureau of Transport and Regional Economics
CARFG	California Reformulated Gasoline
CEC	California Energy Commission
CIDI	Compression-Ignition, Direct-Injection
CMG	Compressed Methane (biogas)
CNG	Compressed Natural Gas
CO	Carbon Monoxide
CO_2	Carbon Dioxide
CONV	Conventional
CUEDC	Composite Urban Emissions Drive Cycle
DEH	Department of the Environment and Heritage
DI	Direct Injection
DIPE	Di-isopropyl Ether
DNRE	Department of Natural Resources and Environment
DOTARS	Department of Transport and Regional Services
DSA	Duncan Seddon and Associates
DTRD	Department of Transport and Regional Development
E10	Petrol containing 10% ethanol by volume
EA	Environment Australia
EF	Emission Factor
EGCS	Energy Grants Credit Scheme
EGR	Exhaust Gas Recirculation
EPA	Environmental Protection Agency
ETAE	Ethyl Tertiary Amyl Ether
ETBE	Ethyl Tertiary Butyl Ether
EtOH	Ethanol
EV	Electric Vehicle
FC	Fuel Consumption
FFC	Full Fuel-Cycle
FTD	Fischer-Tropsch Diesel
FTP	Federal Test Procedure
GHG	Greenhouse Gas
GGAP	Greenhouse Gas Abatement Program
GI	Grid-independent (charge-sustaining)
GM	General Motors Corporation

GREET	Greenhouse Gases, Regulated Emissions, and Energy use in Transportation
GWP	Global Warming Potential
H_2	Hydrogen
HEV	Hybrid Electric Vehicle
HGV	Heavy Goods Vehicle
HHV	High Heating Value
HV	Heavy Vehicle
ICE	Internal Combustion Engine
IDF	Industrial Diesel Fuel
IPA	Iso-propyl Alcohol
IPCC	Intergovernmental Panel on Climate Change
kt	Thousand Tonnes
LCA	Life-Cycle Analysis
LCV	Light Commercial Vehicle
LDV	Light Duty Vehicle
LHV	Low Heating Value
LLSP	Laser Light Scattering Photometry method to measure PM
LNG	Liquefied Natural Gas–primarily propane with small amounts of butane and iso-butane
LPG	Liquefied Petroleum Gas
LSD	Low Sulfur Diesel ($S \leq 500$ ppm)
LP	Leaded petrol
LT	Long-term
LTIS	Long-term in service
MeOH	Methanol
MGV	Medium Goods Vehicle
MIR	Maximum Incremental Reactivity
MON	Motor Octane Number
Mt	Million Tonnes
MTA	Automated Manual Powertrain
MTBE	Methyl Tertiary Butyl Ether
MVEC	Motor Vehicle Environment Committee
N_2O	Nitrous Oxide
NAFC	National Average Fuel Consumption
NEPC	National Environment Protection Council
NG	Natural Gas
NGGIC	National Greenhouse Gas Inventory Committee
NICNAS	National Industrial Chemical Notification and Assessment Scheme
NMVOC/NMOC	Non-Methane Volatile Organic Compounds/Non-Methane Organic Compounds
NRC	National Research Council
NO_x	Nitrogen Oxides
NSTC	National Science and Technology Council
NT	Near-term
O_2	Oxygen
OFP	Ozone Forming Potential
ORNL	Oak Ridge National Laboratory
PAN	Peroxyacetyl nitrate
PAH	Polycyclic Aromatic Hydrocarbons
PM	Particulate Matter
PM10	Fine particulates with a mean aerodynamic diameter of 10 µm or less
ppm	Parts per million
PULP	Premium unleaded petrol
RAF	Reactivity Adjustment Factor

RFG	Re-formulated gasoline
RON	Research Octane Number
RME	Rapeseed Methyl Ester
RVP	Reid Vapour Pressure
SHED	Sealed Housing for Evaporative Determination
SIDI	Spark-Ignition, Direct-Injection
SMR	Steam Reforming
SO_2	Sulfur Dioxide
SO_x	Sulfur Oxides
TAC	Toxic Air Contaminants
TAME	Tertiary amyl methyl ether
TBA	Tertiary butyl alcohol
TEOM	Tapered Element Oscillating Microbalance (instrument to measure PM)
(T)HC	(Total) Hydrocarbons
TTW	Tank to Wheel
TWC	Three way catalyst
UDDS	Urban Driving Dynamometer Schedule
ULP	Unleaded petrol
ULSD	Ultra-Low Sulfur Diesel ($S < 50$ ppm)
VATS	Victoria Activity and Travel Survey
VOC	Volatile Organic Compounds (see HC)
v/v	Volume per volume
WTT	Well to Tank
WTW	Well to Wheel
XLSD	Extra Low Sulfur Diesel ($S < 10$ ppm)

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HIGHLIGHTS

….. a biofuels industry would require substantial and ongoing assistance

- ⊡ The production of ethanol and biodiesel for transport fuel use in Australia is currently around 60 ML and is projected to increase to 115 ML by 2010 (with the phase-out of the effective excise relief, as assumed in this study).
  - … Underlying growth in biofuels production is expected to be sourced from additional ethanol produced from waste starch using existing capacity (30 ML) and new biodiesel produced from waste cooking oil (around 30 ML).
  - … Both of these options are considered to be cost competitive with traditional fuels over the medium to longer term, even in the absence of government assistance and where they are taxed on a comparable basis with other fuels.
- ⊡ However, in order to reach a target of 350 ML in 2010, the biofuels industry would require *substantial and ongoing* assistance.
  - … In the absence of government assistance, all other options for producing biofuels in Australia examined in this review are considered unlikely to be cost competitive with traditional fuels over the medium to longer term.
- ⊡ Under a scenario where the 350 ML biofuels target is assumed to be met, the additional 235 ML of biofuels is assumed to comprise 205 ML of ethanol and 30 ML of biodiesel.
  - … The ethanol is assumed to be produced using C molasses (60 ML) and whole cereal grains (145 ML).
  - … The biodiesel is assumed to be produced from waste cooking oil (another 30 ML).

….. with the biofuels target, GHG emissions are reduced by 268,000 tonnes

- ⊡ Under the target scenario, greenhouse gas emissions are estimated to be 268,000 tonnes lower in 2010 (about 0.3 per cent of transport GHG emissions).
  - … Total costs in 2010 (in terms of lost GDP) are estimated to be between \$265 to \$277 a tonne CO_2-e (in 2003 dollars).
  - … The cost to government (in 2010) is estimated to be between \$113 to \$163 a tonne CO_2-e (in 2003 dollars).
  - … In comparison, the upper bound of the cost to government of abatement purchased under round 1 of the Greenhouse Gas Abatement Program is around \$10 a tonne. The abatement relates to the period 2008-2012 (the first commitment period of the Kyoto Protocol).
  - … Sensitivity Analysis shows that, under a ‘best case’ scenario, where all the ethanol is assumed to be produced from C molasses using bagasse fuelled cogeneration, greenhouse gas emissions could be reduced by up to 360,000 tonnes.

….. other environmental impacts are small

- ⊡ Other land, water and biodiversity impacts, from the production, distribution and use of 350 ML of biofuels are not significant, provided wastes are disposed of appropriately.

….. savings in health costs equal \$3.3 million

- ⊡ Savings in health costs of meeting a biofuel target are estimated to be \$3.3 million in 2010 (in 2003 dollars).

… Savings in health costs from increased ethanol use of 205 ML are estimated to be $1.8 million in 2010. These arise mainly from the reduced impact of particulate matter emissions, with the replacement of metropolitan area petrol refining with regionally based ethanol production.

… Savings in health costs from increased biodiesel use of 30 ML are estimated to be $1.5 million in 2010, mainly due to lower tailpipe particle emissions.

… Under a ‘best case’ scenario, savings in health costs are estimated to be $5.2 million in 2010.

… There remain large uncertainties associated with estimates of the impacts on air quality from the use of ethanol, particularly from evaporative emissions.

….. the target reduces GDP by $70 million

⊡ Assisting the biofuels industry to meet a 350 ML target is estimated to reduce GDP by between $71 and $74 million in 2010 (in 2003 dollars).

… This economic loss reflects both reduced efficiency (using more costly transport fuels and bidding resources away from productive activities) as well as the impact of increased taxes or reduced government expenditure on services.

….. ….. direct jobs estimated to cost around $500,000 each

⊡ The cost of each direct job created is is estimated to be between $492,000 and $516,000 (in terms of lower GDP) or from $164,000 to $172,000 for each direct and indirect job created in biofuels and related industries.

⊡ Regional employment impacts of biofuels production are commonly overstated and are difficult to predict as they will be plant and location specific. Some regions will benefit; however, given the mobility of labour, not all of these jobs would represent additional employment. Benefits will be localised and are likely to be concentrated in parts of Queensland and New South Wales.

….. security

⊡ The benefits of biofuels in terms of improving energy security are minimal. 350 ML of biofuels represents only 1.1 per cent of Australia’s total motor vehicle fuel demand.

… As Australia’s energy security position evolves over the longer term (20 to 30 years), it is appropriate that energy policy be developed on an energy sector wide basis, including (among other things) fuel diversification, technology and innovation, renewable energy, energy efficiency and conservation.

….. key assumptions

⊡ The analysis undertaken necessarily includes numerous assumptions.

… The outlook for world oil prices, exchange rates and the prices of biofuel feedstocks is based on ABARE’s most recent medium term outlook.

… The analysis focused on production options that do not require further extensive research and development. Ethanol production from lignocellulosics and the new ZeaChem process are believed to have potential, over the longer term, but the technical viability and economics of each are not yet well established.

….. conclusions

⊡ The costs of implementing a policy of assisting the Australian biofuels industry to meet a 350 ML biofuels target are estimated to exceed the benefits.

… Ethanol produced from waste starch and biodiesel produced from waste cooking oil both appear to be (or are close to being) economically viable without government assistance and should be able to compete effectively in an environment where they are taxed on a comparable basis with other fuels. However in both cases future growth in production is expected to be modest due to the limited availability of feedstock supplies.

… Ethanol produced from molasses and cereal grains and biodiesel produced from tallow or oilseeds all require substantial and ongoing government assistance to be viable.

… Assistance to the biofuel industry would generate some benefits in terms of health (via improvements in air quality), reductions in greenhouse gas emissions and regional employment opportunities. However, in all cases, these benefits were found to be small and varied with the biofuel source, production practices and utilisation circumstances.

Summary of key results

Biofuels industry and viability	
current biofuel use [a]	50-60 ML
projected biofuel use in 2010 [a]	115 ML
Additional production required to meet a 350 ML target	235 ML
contribution from ethanol	205 ML
… using C molasses	60 ML
… using whole cereal grain	145 ML
contribution from biodiesel (using waste cooking oil)	30 ML
Required revenue (production costs less revenue from coproducts) [b]	
ethanol (using existing capacity)	18–26c/L
ethanol (using new capacity)	32–71c/L
biodiesel (using new capacity)	35–119c/L
Prices of competing fuels [c]	
petrol	20c/L
diesel	30c/L
Costs of meeting a 350 ML biofuels target in 2010	
Reduction in GDP in 2010[d]	$70.9 to $74.3 million
Cost of greenhouse gas emission reductions	
cost to government	\$113 to \$163/tonne CO_2-e
total economic cost (loss in GDP)	\$265 to \$277/tonne CO_2-e
Economic cost per job (direct and indirect)	$164,120 to $171,990 per job
cost per direct job	$492,360 to $515,970 per job
Benefits of meeting a 350 ML biofuels target in 2010	
Greenhouse gas emission reductions	268,000 tonnes
from ethanol	184,000 tonnes
from biodiesel	84,000 tonnes
Savings in health costs in 2010	$3.3 million
from increased ethanol use of 205 ML	$1.8 million or 0.9c/L
from increased biodiesel use of 30 ML	$1.5 million or 5.1c/L
Jobs created in biofuels and related industries (direct and indirect)	432
direct jobs in biofuels	144

[a] Only includes biofuels used in the transport fuel market. [b] All prices and cost estimates are quoted in 2003 dollars. [c] Prices of competing fuels are expressed in energy equivalent terms. [d] The range of cost estimates reflects whether all 350 ML of biofuels are subsidised or only the additional 235 ML required to meet the target.

EXECUTIVE SUMMARY

In July 2003, the Australian Government commissioned a report from the Commonwealth Scientific and Industrial Research Organisation, jointly with the Bureau of Transport and Regional Economics and the Australian Bureau of Agricultural and Resource Economics, on the appropriateness of maintaining an objective that biofuels, produced in Australia from renewable resources, contribute at least 350 million litres (ML) to the total fuel supply by 2010. The Government announced the objective in the context of its 2001 re-election campaign.

Using existing literature, the study was to assess the net environmental (ie greenhouse, air quality and other, including health) benefits of replacing fossil fuels with Australian sourced biofuels in the Australian transport mix, together with the economic and regional benefits. The study was to include consideration of industry viability in producing biofuels in Australia and the cost-effectiveness of assistance to achieve any estimated net public benefits.

In light of an effective six to eight year time horizon for the study, the focus is on production options that are already technically viable, or very close to being so, rather than on alternatives that may possibly become viable in a longer time frame. Thus, for example, ethanol production from lignocellulosics and the new Zeachem process are not included in the analysis.

THE BIOFUELS INDUSTRY

Biofuels currently provide around 50 to 60 ML (0.3 per cent) of road transport fuel. Ethanol from wheat starch, produced by Manildra at its Nowra, New South Wales plant and used principally as a petrol additive in a blended fuel (E10), is the source for around 50 ML. About 5 ML of ethanol is produced by CSR from C molasses feedstock at the Sarina (Queensland)/Yarraville (Melbourne) facilities.

While it is difficult to gauge biodiesel production precisely, 2002-2003 production is estimated at around 0 to 2 ML. Biodiesel production using canola feedstock was estimated at 23 ML in 2002, but appears to have fallen away subsequently, with high, drought-affected prices for canola oil. Biofuel Industries Australia has recently opened a biodiesel plant using waste oil at Rutherford, New South Wales, with a capacity of 14 to 17 ML. With some other facilities also expected to commence, production appears likely to increase somewhat over the near term. Biodiesel (BD100) is sold primarily as a diesel alternative for heavy vehicles in a small number of independent service stations.

POLICY CONTEXT

Biofuels production to 2010 and beyond will be influenced by Australian Government policy settings.

Fuel tax reform

Ethanol and biodiesel are currently effectively untaxed. In the 2003–2004 budget, the Government announced reform of current fuel tax arrangements to bring all currently untaxed fuels used in internal combustion engines into the excise and duty system from 1 July 2008. Final rates, to be phased in and to apply from 1 July 2012, announced on 16 December 2003 take account of factors including the energy content of fuels. These changes will have a significant impact on the long-term viability of ethanol and biodiesel production.

Fuel quality standards

National fuel quality standards, under the *Fuel Quality Standards Act 2000*, are designed to reduce the level of pollutants and emissions from the use of fuel, improve vehicle operation and facilitate the introduction of the new vehicle emission control technologies required to achieve the tighter emission standards. Fuel standards for petrol and diesel are to become progressively tighter over the period to 2006 and this trend is expected to continue thereafter. Under the current proposed scenario, 50 parts per million sulfur premium unleaded petrol (PULP) will be introduced by 1 January 2008, with 10 parts per million sulfur PULP to be in place by 1 January 2010 (reduced from 150 parts per million sulfur currently). Similarly, extra low sulfur (10 parts per million) diesel would be introduced by 1 January 2009 (reduced from 500 parts per million today and from 50 parts per million from 1 January 2006, as mandated).

Thus it is important to measure the net environmental impact of increased biofuels use against the emissions performance of the vehicle fleet, as it is likely to be at the end of the decade, not as it is today.

Higher octane petrol and ethanol

Ethanol can be used both as a fuel ‘extender’, where a blend of ethanol is added to petrol (as in 10 per cent ethanol petrol, E10), and as an oxygenate and octane enhancer. The post-2006 petrol vehicles standards will be mandated with reference to higher octane petrol (i.e. 95 RON PULP), with a view to aiding progress towards an 18 per cent improvement in the fuel efficiency of new vehicles by 2010, to 6.8 litres per 100 kilometres travelled, as per the National Average Fuel Consumption target.

In principle, this standard may improve the future viability of ethanol, either by marginally increasing the production cost of petrol, or by creating increased opportunities to use ethanol as an octane enhancer. In addition, methyl tertiary butyl ether (known as MTBE), currently the most widely used fuel oxygenate, will be effectively banned in Australia from 1 January 2004, due to its high potential to contaminate groundwater.

Specific biofuels measures

To encourage new entrants to the biofuels industry, projects that provide at least 5 million litres in new or expanded biofuels capacity will be eligible for a subsidy of 16c/L of capacity (to a maximum of $10 million per project). Applications for the subsidy close during January 2004.

During 2003, a 10 per cent ethanol limit in petrol and a biodiesel standard have come into force. The measures can be expected to help build or, where necessary, rebuild community confidence in the fuels. An ethanol fuel labelling standard was made and will come into effect from 1 March 2004. An ethanol fuel quality standard is under consideration.

VIABILITY OF BIOFUELS FOR TRANSPORT

Viability of ethanol

Industry viability is assessed on the basis of whether or not biofuels production is considered likely to be cost competitive with traditional fuels over the medium to longer term; that is, after the period 2008 to 2012, when current government assistance arrangements are set to be phased out. In order to compete with petrol in the absence of current production and capital grants, ethanol must be able to meet a threshold price.

This threshold price is principally determined by the world market price of oil, the exchange rate, the applicable taxation and/or subsidy regime and the relative energy content of the ethanol (assumed to be 68 per cent). While oil prices have been at historically high levels in recent years, some easing of prices is expected in the near term, with further easing in the period 2008 to 2012, to around US$23 a barrel (in 2003 dollars) in West Texas Intermediate terms. The Australian dollar is projected to return, over time, to a trend level of US 65 cents. With an assumed medium term cost of petrol production in Australia of 29c/L (in 2003 dollars), the medium to longer-term threshold price for ethanol (in the absence of current production and capital grants), is estimated to be 20c/L (again in 2003 dollars). In the longer term, with higher-octane petrol expected to become standard, the threshold price could potentially increase to around 23c/L.

Feedstock is the major cost in producing ethanol (and biodiesel). Because of this, the initial development of these industries in Australia has been focused around the supply of low cost waste products or by-products. The study examined the costs of production of ethanol from: waste starch using existing capacity; C molasses using existing capacity; C, B and A molasses, each using new capacity; and cereal grains (degraded wheat or sorghum) using new capacity.

The break-even revenue required for the production of ethanol from waste starch (using existing capacity) is estimated to be below the medium term threshold price of 20c/L of ethanol. However, waste starch is a limited resource and investment in new production facilities based on this feedstock is unlikely in the near future. Capital costs, for new ethanol production facilities from all feedstocks, are conservatively estimated at between 7 and 9c/L. For new investments that qualify for a capital grant, these fixed capital costs could be reduced by up to 1c/L (depending on the plant size).

Ethanol produced from whole cereal grains generates crushed grain meal (or distiller's grain), a valuable by-product for use as livestock feed. Taking this into account, the net required revenue for ethanol production based on using cereal grains (and new capacity) is estimated to be 32c/L of ethanol. This is significantly higher than the required threshold price of 20c/L.

The cost of ethanol produced using C molasses feedstock varies significantly through time and is influenced significantly by the supply of and demand for exported raw sugar. The net required revenue for ethanol production based on using C molasses is estimated at 26c/L of ethanol (existing capacity) and 33 cents (new capacity). Again this is significantly higher than the required threshold price of 20c/L. In the case of ethanol produced from B and A molasses feedstocks, while the energy yield is greater, this does not compensate for the increased cost of feedstock and neither of these production options is considered to be economically viable.

Viability of biodiesel

Biodiesel can be used as an extender for diesel fuels, but also as a replacement, as 100 per cent biodiesel fuel (BD100) can be used in existing diesel engines with minor modification. Biodiesel production also generates glycerin (glycerol or glycerine), a valuable by-product commonly used as a solvent, plasticiser and softening agent in a wide range of industries.

With biodiesel assumed for the purposes of analysis to have 90 per cent of the energy content of diesel, the study estimates that biodiesel will need to compete with a medium term cost of production of diesel in Australia of 30c/L (in 2003 dollars). The production options examined in this report are biodiesel produced (using new capacity) from: waste cooking oil; tallow; oilseeds, and canola oil.

As with waste starch, it is difficult to determine the true economic value of waste cooking oil, as no transparent market exists. Some businesses incur costs associated with the disposal of the waste oil, while others are paid for it. The Australian Taxation Office has estimated the price of waste cooking oil at $170 a tonne, or 20c/L. At this assumed price and assuming a glycerin by-product revenue stream (6c/L), the estimated cost of biodiesel production using waste cooking oil feedstock is 35c/L (in 2003 dollars). Given the degree of uncertainty surrounding many of the cost items included in the analysis, this difference is too small to conclude that biodiesel production based on using waste cooking oil feedstock is not viable over the medium to long term. It is possible that increased awareness in the food industry of the market value of waste oil could lead to an increase in supplies, as product is diverted away from current disposal methods. While potential stocks are ultimately constrained by the level of industry and household demand for cooking oil and by accessibility considerations, such a change could lead to a fall in the price of waste cooking oil over the medium term and to further growth in the industry.

In the case of biodiesel produced from tallow, the net revenue required to cover costs is estimated to be 66c/L (in 2003 dollars). Corresponding figures from biodiesel produced from whole grain oil seeds and canola oil are 76c/L and $1.19 a litre respectively. Given these results, none of these options are considered to be economically viable.

BIOFUELS AND THE ENVIRONMENT

Biofuels and air pollution

The air quality implications of biofuels depend on the exact nature of the vehicle in which the fuel is used, the exact nature of the fuel with which the biofuel is blended, and the exact nature of the airshed into which the exhaust and evaporative pollutants are emitted. In addition, the difficulty of extrapolating from individual vehicle test results (undertaken on dynamometers) to in-service conditions means that there are large uncertainties associated with any estimates of the air quality implications of the use of biofuels.

The use of E10 leads to increased aldehyde emissions from tailpipe emissions, but decreased benzene and 1,3-butadiene emissions from tailpipes. When these are combined into an air toxics index, the use of E10 reduces the value of the index (on a per kilometre basis) by 17 per cent.

There is still insufficient knowledge about evaporative (i.e. pre-combustion), as opposed to tailpipe, emissions, to make a clear judgement on the impacts of E10 on ozone formation. Reactive hydrocarbons (also known as volatile organic compounds) in the atmosphere combine with oxides of nitrogen and sunlight to form ozone, the major constituent of smog. Some of these compounds are also air toxics. Lack of knowledge of the detailed composition of evaporative emissions in terms of both ozone precursors and air toxics adds yet another uncertainty to the estimates of the air pollution potential of E10.

The likely health impact of biofuels is influenced strongly by the emissions of particulate matter, which recent epidemiological research finds is closely linked to both respiratory disease and mortality outcomes of those exposed to it. On a fuel life cycle basis, unless energy is supplied through cogeneration, ethanol (as E10) emits slightly more particles than petrol, especially if coal is used to provide energy in the milling and distilling stages. If gas rather than coal based energy resources were used, there would be a smaller increase in particle emissions, while, with cogeneration, particle emissions would fall.

Biodiesel emits far fewer particles than low sulfur diesel (i.e. up to 38 per cent fewer life cycle emissions). The decrease is somewhat less in comparison with extra low sulfur diesel (about 20 per cent), which is expected to become standard by 2010.

There is insufficient data at the present time to assess the air toxic emissions from biodiesel.

Biofuels and greenhouse

On a fuel life cycle basis, there can be greenhouse gas savings of up to 5 per cent from the use of E10. In contrast, the use of 100 per cent biodiesel made from waste oil can achieve 90 per cent greenhouse gas benefits compared with diesel because a waste product does not have any life cycle emissions (also known as “exbodied”, or “well to wheel” emissions) associated with it.

Other impacts

Other land, water and biodiversity impacts, from production, distribution and use of biofuels appear not to be significant, provided that distillery wastes are disposed of using established best practice.

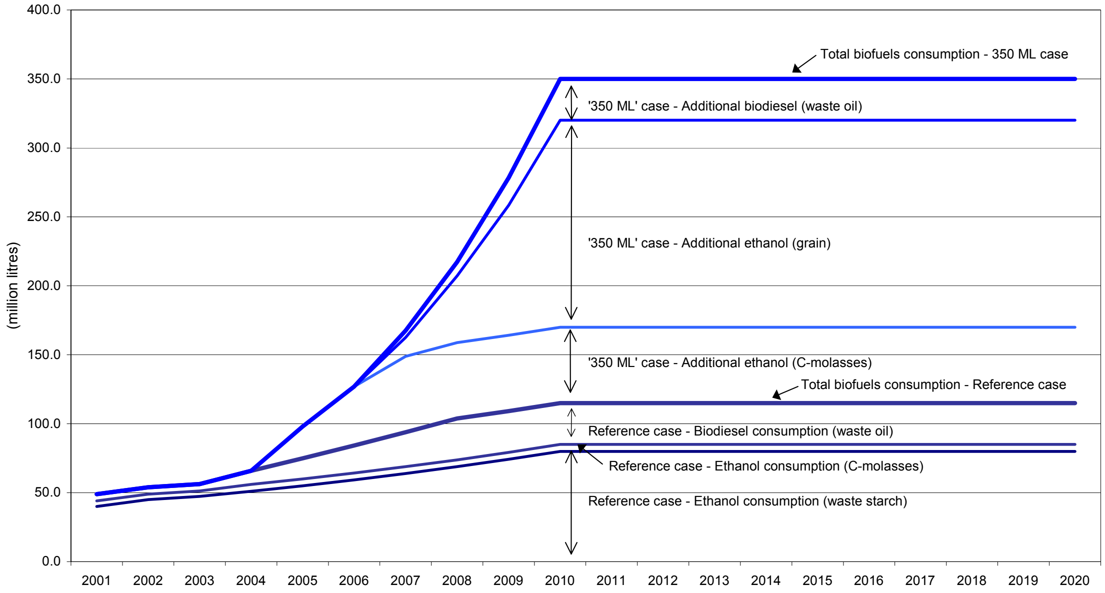
TRANSPORT EMISSIONS AND A 350 MILLION LITRE TARGET

Reference and 350 million litre cases

The study estimates that, under existing policy settings, and assuming that future final fuel excise rates based on fuel energy content are confirmed, biofuels use in transport could increase from around 50–60 ML currently to around 115 ML in 2010. The reference case scenario (Figure 1) could comprise 80 ML of ethanol from waste starch, using existing production capacity, 5 ML of ethanol from C molasses, as at present and 30 ML of biodiesel from waste oil sources (mainly new projects).

The reference case assumes that the recent community unease regarding ethanol use in motor vehicles abates. It further assumes that demand for ethanol as an octane enhancer does not increase later in the decade. There will be a range of alternative strategies for the refining industry to choose between, including relying largely on additional refining. Cost competitiveness, volatility (in summer months) and reliability of supply would appear to be threshold challenges for ethanol as an octane enhancer.

Figure 1. Projected biofuels mix: Reference case and 350 ML biofuels consumption case



Thus reaching 350 ML would involve use of a net additional 235 ML of biofuels. Tax relief or subsidy measures would be necessary for this replacement production to be economically viable.

The viability analysis suggests that, on a least cost basis, the 235 ML could comprise a further 30 ML of biodiesel from waste oil and 205 ML of ethanol (145 ML sourced from damaged grains and 60 ML from C molasses). This would replace an energy equivalent volume of diesel (27 ML) and petrol (140 ML).

Change in emissions from 350 million litres of biofuels

The increased biofuels consumption would result in decreased carbon monoxide emissions, but increases in overall nitrogen oxides, volatile organic compounds (principally, hydrocarbons) and particulate matter emissions. Most of the reduction in carbon monoxide emissions is attributable to the lower carbon monoxide tailpipe emissions of E10 blend fuels relative to unleaded petrol. The projected increases in nitrogen oxides result because ethanol blends produce higher nitrogen oxides than petrol. The higher output of total volatile organic compounds is the net result of two effects: a reduction in exhaust emissions, but an increase in evaporative emissions, due to the higher fuel volatility of E10. Total particle emissions are also projected to increase, largely because of a substantial particulate matter penalty assumed with production of ethanol with electricity produced from coal-fired power stations.

The increased consumption of biofuels would result in reduced greenhouse emissions, of approximately 0.27 million tonnes in 2010. Most of the reduction stems from the fact that ethanol and biodiesel are produced from renewable sources. By convention, greenhouse emissions generated as a result of combustion of a fuel produced from a renewable source are not included when calculating total carbon dioxide equivalent emissions. Upstream emissions, however, are projected to increase, due to the more greenhouse intensive character of ethanol production.

HEALTH IMPACT

Epidemiological studies have shown a close link between concentrations of toxic substances in urban air sheds and morbidity and mortality rates amongst residents. To assess the impact on health that the emissions profile from 350 ML of biofuels use would have, it is necessary to make assumptions about the locations of production and use of the fuels (and of those that they would replace).

Increased ethanol use

It is assumed that the extra ethanol production would take place in three separate rural locations: northern New South Wales, southwest Queensland (both grain feedstock) and northern Queensland (molasses feedstock). The extra biodiesel production is assumed to occur on the fringe of a large metropolitan area, in reasonable proximity to sufficiently sized sources of the waste oil feedstock. Most of the transport use, for both fuels, is assumed to occur in major metropolitan areas.

Using a recent set of Australian health impact values,[1] which incorporates monetary estimates of loss of life (i.e. impact on mortality) and lost quality of life (i.e. impact on morbidity), as well as medical system costs, the change in total life cycle pollutant emissions from an extra 205 ML of ethanol in 2010 would result in some small health benefits ($1.8 million, or approximately 0.9c/L of additional ethanol supply). These savings are mostly attributable to reduced pollutant emissions in urban areas, primarily from reduced production of petrol at refineries based in metropolitan areas. This effect greatly outweighs any increase in particulate matter emissions from the replacement ethanol production, since the latter would occur in regional locations, where total health impacts will be less.

The beneficial health impact arising from reduced carbon monoxide, volatile organic compound and particulate matter tailpipe emissions is partially offset by increased oxides of nitrogen. Because of the uncertainty surrounding the toxic composition of E10 evaporative emissions, the study did not attempt to cost the health impact of the likely increase in evaporative emissions occasioned by increased E10 use.

Increased biodiesel use

The savings in health costs from an additional 30 ML of biodiesel consumption are estimated to total $1.5 million from reduced tailpipe and net upstream emissions, combined. A lower tailpipe emission rate of particles, even by comparison with extra low sulfur diesel, is the main contributing factor. Average benefits would be around 5c/L of additional biodiesel supply.

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[1] P. Watkiss (2002), Fuel Taxation Inquiry: The Air Pollution Costs of Transport in Australia, AEA Technology Environment.

ALTERNATIVE SCENARIOS

Alternative energy sources for ethanol production

The viability analysis showed only a 1c/L cost differential between ethanol sourced from whole cereal grains and ethanol produced from C molasses. For the additional 205 ML to be sourced from C molasses, sugar mills would need to develop suitable storage facilities to enable year-round ethanol distillery operations, at some undefined additional cost. Should such production eventuate, the power source would be cogeneration in sugar mills, based on year-round bagasse supply. As a result, total particulate matter emissions from the additional ethanol use would fall rather than increase, with estimated health cost savings of $3.7 million. Greenhouse gas emissions would fall by 277,000 tonnes in 2010.

Ethanol production using bagasse cogeneration represents, in effect, a best case, in terms of environmental impact. In contrast, ethanol from coal-fired boilers and coal-fired electricity, currently the principal energy sources, constitutes a worst case. Gas-fired steam generation is a potential option in producing ethanol. Its environmental impact will fall in the range between those of these other two energy sources.

Emissions and health cost uncertainties

Uncertainty analysis of the difference in emissions performance between biofuels and the reference fuels suggests that, while there is an 82 per cent probability that health costs will reduce with increased ethanol use, there is some probability (18 per cent) that health costs will increase slightly. This reflects a relatively wide range of test results for the impact of E10 on tailpipe particulate matter emissions. There is also a small (11 per cent) probability that greenhouse gas emissions will increase, rather than reduce, from use of ethanol.

Changes in emission rates of biodiesel, notwithstanding the significant uncertainties involved, would not alter the finding that health costs are expected to reduce.

REGIONAL IMPACT

The regional employment impacts of biofuels production have commonly been overstated and are difficult to predict, as they will be plant-specific and location-specific. Potential benefits include: stimulus to the local agricultural sector; increased employment, in terms of direct jobs in the plant and an indirect or flow-on effect to the local economy; and revitalisation of rural communities. Potential costs include: higher costs for industries competing for the same inputs, such as the livestock feed industry; and diversion of product away from exports towards subsidised domestic production.

Regional benefits are likely to arise only from those projects located in non-urban areas, specifically ethanol plants. Viable biodiesel plants, based on waste oil sources are less likely to offer regional development benefits. If non-urban ethanol plants do proceed, some regional benefits will certainly result, particularly in terms of employment. Such benefits will be localised and are likely to be concentrated in parts of Queensland and New South Wales.

To calculate possible employment impacts, an estimate by proponents that each new biofuel plant of 60 ML capacity would generate 36 direct jobs is adopted. Four such additional plants would be required to produce an additional 235 ML of biofuel, producing 144 jobs. On the further assumption that each direct job leads to two additional indirect jobs, total resulting employment could amount to a maximum of 432 jobs. Not all of these jobs would represent additional employment. Labour, particularly skilled labour, is a mobile resource within the economy and development of a biofuels industry could be expected to compete resources away from other sectors and regions, notwithstanding the existence of any unemployment in areas where the plants would be located.

The estimated subsidy required to induce sufficient investment to meet the 350 ML target is equivalent to government expenditure per direct job (in 2010) of between \$210,000 and \$303,000, with the expenditure per both direct and indirect jobs estimated to be in the range \$70,000 to \$101,000.

ECONOMIC IMPLICATIONS

Impact on GDP

In the case where the production of an additional 235 ML of biofuels is subsidised (to meet the 350 ML target), gross domestic product (GDP) would be \$71 million lower in 2010 (\$74 million if assistance were provided to all 350 ML). This economic loss reflects both reduced efficiency (using more costly transport fuels and bidding resources away from productive activities) as well as the impact of increased taxes or reduced government expenditure on services, which is required to fund the subsidy.

Taking into account the production build-up that would be necessary to meet a 350 ML target in 2010, the total net present value of costs to the Australian economy from 2004 to 2010 is estimated to be between \$95 million and \$100 million.

The estimate of the annual economic welfare gain from avoided health impacts associated with increased biofuels use is \$3.3 million. This is akin to an increase in the asset or human capital base (stock) of the economy, whereas GDP is an income (flow) measure. While the avoided health cost estimate implies that there would be a small offsetting positive impact on GDP, it is not appropriate to subtract this estimate from the estimate of loss in GDP.

Valuing greenhouse and health benefits

With the additional biofuels use reducing greenhouse emissions by an estimated 268,000 tonnes, the cost per tonne of carbon dioxide equivalent emissions (in terms of lost GDP) is estimated to be between \$265 and \$277 per tonne CO_2-e.

In the absence, at the present time, of an international market value for carbon dioxide equivalent emissions, the Australian Greenhouse Office has suggested use of values contained in its 1999 publication, *Discussion Paper 2 – Issuing the Permits*. The discussion paper postulated a permit price range of \$10 to \$50 a tonne. The lower value of \$10 a tonne is consistent with the upper bound of the cost to government of abatement purchased under round 1 of the Greenhouse Gas Abatement Program (GGAP). The abatement purchased under GGAP relates to the period 2008-2012, which is the first commitment period under the Kyoto Protocol. If a value of \$10 a tonne is used, then the implied value of greenhouse gas abatement from the additional biofuels use would be \$2.7 million, or 1.1c/L.

Biofuels and energy security

An important element of energy security is fuel self-sufficiency, which may be improved by (among other things) increased use of non-oil fuels, including renewables, by improved fuel efficiency and energy conservation. However, achieving a 350 ML biofuels objective would replace only 1.1 per cent of Australia's total motor vehicle fuel demand. This is too small an amount to make a material contribution to greater energy security for Australia. Moreover, achieving a higher target, at greater economic cost, appears unlikely to be a cost-effective energy security strategy.

Economic cost of biofuels employment

It is estimated that the economic cost (in terms of lower GDP) of each biofuels and related job (in 2010) is between $164,000 and $172,000 (in 2003 dollars). In the case of direct jobs created, the economic cost for each job in 2010 is estimated to be between $492,000 and $516,000 (again in 2003 dollars). The range of costs reflects whether subsidies are applied to the whole 350 ML of production, or only the additional 235 ML required to meet the target.

1 TERMS OF REFERENCE

On 25 July 2003 the Australian Government announced that it would commission a report on the objective, set in 2001, that biofuels woiuld contribute at least 350 ML to the total fuel supply by 2010. The final terms of reference agreed for the study are as follows:

The Commonwealth Scientific and Industrial Research Organisation, jointly with the Bureau of Transport and Regional Economics and the Australian Bureau of Agricultural and Resource Economics (the parties) investigate the appropriateness of maintaining an objective that biofuels, produced in Australia from renewable resources, contribute at least 350 ML to the total fuel supply by 2010.

Using existing literature, studies and scientific reports, including from overseas, assess the net environmental (greenhouse, air quality and other including health), economic and regional benefits of replacing fossil fuels with Australian sourced biofuels in the Australian transport mix.

In making this assessment, the parties should have regard to:

- industry viability (technical feasibility and economic viability) of producing biofuels in Australia, currently and in the longer term, taking into account announced changes to excise arrangements, including the phase-out of effective excise relief and the final excise rates applying from July 2012 (to be announced later this year), and taking account of current and planned changes in fuel standards, including the introduction of an E10 ethanol standard;

- the effect of current levels of government assistance for biofuels (excise relief, capital subsidies, GGAP grants) on biofuels production including the level of production that could be expected based on these measures;

- the net environmental (greenhouse, air quality and other including health) impacts of replacing fossil fuels with biofuels in the Australian transport mix;

- the net regional effects of replacing fossil fuels with biofuels in the Australian transport mix;

- the cost effectiveness of assistance to achieve any estimated net public benefits.

The report by the parties should be presented to Government no later than end November 2003.

2 SCOPE OF WORK

The tasks undertaken to meet the terms of reference included:

- collation of all relevant Australian and international reports / studies;
- a future fuel use projection to 2020 that covers both the traditional fuels (petrol and diesel) and the biofuels (ethanol plus biodiesel);
- projected oil prices to 2020 and the relative price of biofuels.

This information was used to identify whether there are physical limitations (in contrast to economic limitations) to achieving the 350 ML biofuels target, and whether such limitations, if they exist, impact more significantly on ethanol, than on other fuels such as biodiesel. This information was also needed to determine the key issues / options / uncertainties. For example, the pattern of use (e.g. urban versus rural) needs to be assumed and is likely to impact significantly on, at least, regional benefits and on air quality considerations.

For each biofuel, the following is required:

1. Scenarios of the likely future ethanol producers, their source of feedstock, and the location of their facilities.
2. Estimates of life cycle production and distribution costs and market demand.
3. The GHG/criteria pollutant/air toxics emissions and other external benefits and costs associated with production and distribution.

Vehicle use

4. Estimates of the recent literature on emissions of GHG/criteria pollutant/air toxics from:
    i) light vehicles using ethanol as a 10% (by volume) blend in petrol; and
    ii) heavy vehicles using biodiesel.

Contributions from Partners in the Review

CSIRO coordinated the review and was responsible for securing agreement amongst the review partners to the review on the scope and allocation of responsibilities. CSIRO was also responsible for liaison with the Interdepartmental Committee (via Marie Taylor).

CSIRO's input was:

- lead role in assessing the net environmental benefits issue (this included full fuel lifecycle analyses, as well as waste disposal, land and water, impacts of ethanol and biodiesel enterprises);
- lead role in uncertainty and sensitivity analyses;
- support role in assessing economic viability of biofuels enterprises;
- support role in assessing regional benefits;
- vehicle emissions review.

CSIRO also contributed to issues associated with fuel standards.

ABARE took:

- lead role in assessing industry viability;
- lead role in assessing net economic benefits and other economy-wide considerations;
- support role in assessing regional benefits (particularly in regard to the broader implications of regional support and development);
- support role in assessing net environmental benefits (particularly with respect to greenhouse issues);
- support role in the transport sector fuel consumption projections.

BTRE took:

- lead role in projecting future vehicle fleet transport emissions in light of fuel standards and market developments;
- lead role in assessing regional benefits;
- support role in assessing net environmental benefits;
- support role in assessing net economic benefit and other economy-wide considerations.

The Study was initiated at a meeting of the reference group (later formalised as the interdepartmental steering committee) held on 5 August 2003, with a requirement that the preface be completed by 19 November 2003 and the report delivered by 20 November 2003. A draft report was submitted by that date, with the final report delivered on 19 December 2003.

3 BACKGROUND TO THE REVIEW

The aim is to investigate the "appropriateness" of setting an objective that biofuels (ethanol plus biodiesel), produced in Australia from renewable resources, contribute at least 350 ML to the total fuel supply by 2010.

"Appropriateness" is to be considered in terms of net environmental benefits, net economic benefits, net regional benefits and industry viability:

- net environmental benefits relate primarily to greenhouse gas emissions and air quality impacts associated with biofuels use. They also include "other" sustainability issues associated with a biofuels industry such as waste disposal as well as impacts on land, and water resources;

- net economic benefits refer to the economy-wide effects of government support to achieve the target, taking account of all direct and indirect implications of government support, including the impact on other industries, terms of trade, as well as environmental and regional externalities (where appropriate);

- net regional benefits include economic activity in rural and regional Australia leading to community benefits. The analysis of regional benefits needs to be considered in a broad context, with due consideration being given to the economy-wide implications of specific regional developments such as whether the benefits to one region are achieved at the expense of those of another;

- industry viability is interpreted primarily as the likelihood that a biofuels industry would be cost competitive with traditional fuels at some point between 2008 – 2012 when government assistance to biofuels production is to be phased out.

This analysis proceeded on the understanding that:

- government policy in terms of capital subsidies, production subsidies and import tariffs for the biofuels industry will operate as currently proposed;
- proposed changes in fuel standards in Australia, including the introduction of an E10 ethanol standard proceed as currently planned; and
- the analyses would focus on existing technologies in biofuels production and utilisation.

This report is restricted to only those biofuels that are technically viable. Technical viability refers to the availability and reliability of relevant technologies associated with production and use, suitable feedstocks, implications for waste disposal, etc. Production of ethanol from sugar or starch and biodiesel from waste oil and tallow do not appear to pose any technological problems. However, there are issues with respect to the technical viability (in Australia) of ligno-cellulose based ethanol.

Thus, the report focuses on situations that do not require modification to present infrastructure or more than minor modifications to vehicle technology. Importantly, this means that fuel production or utilisation options that require new research and development are not considered by this review.

4 APPROACH TO THE REVIEW

The scope of a full review of all the potential biofuels, biofuels production processes and utilisation options is too large to be feasible. Given that economic viability and net environmental benefits are likely to be highly conditional on the specific biofuels under consideration, there is a need to focus the review on a "short list" of biofuels that are most relevant to Australia over the next 10 years. Hence, provided here is an overview of the range of biofuels potentially relevant to Australia in the short term.

The selection criteria are based on options that are currently technically viable and (if necessary) on:

(a) potential to contribute to the 350 ML target by 2010 and
(b) likely costs of production relative to traditional fossil fuels (e.g., in A$ per unit of effective energy delivered upon consumption).

On this basis the most significant production processes (on the basis of technical viability, supply and cost considerations) are:

- ethanol from waste streams in the grain processing industry;
- ethanol from sugarcane molasses;
- ethanol from whole-grain sources;
- biodiesel from waste oil streams;
- biodiesel from tallow;
- biodiesel from canola oil.

Two biofuel production and usage scenarios are examined in this study: reference case ('business as usual') and a 350 ML biofuel scenario. The market viability analysis of domestically produced biofuels is used to inform the reference case level of biofuels production and consumption, and also the likely mix of biofuels that would be produced in order to obtain 350 ML of biofuels use in road transport in 2010.

The issues that have been covered in this analysis incorporate:

- specification of the enterprise (product, production process, by-products, utilisation in the marketplace);
- government assistance;
- net environmental benefits (greenhouse emissions on a life-cycle basis, air quality considerations, other environmental issues);
- industry viability (analysis of cost competitiveness and key factors in the production process, input costs, output prices, policy assumptions, technology assumptions etc);
- net regional impacts (economic activity, employment impacts, other community costs and benefits);
- net economic benefits;
- key uncertainties and assumptions;
- conclusions;

as well as issues addressing:

- broader economic issues, inter-industry impacts, trade implications, etc where appropriate;
- international studies and experience (i.e. compare and contrast with the Australian analyses).

The final two chapters include consideration of the cost-effectiveness of government assistance, as well as conclusions.

Greenhouse Gas Emissions

Under the international agreements in relation to carbon accounting, any carbon dioxide that is emitted as the result of the combustion of a renewable fuel is not treated as a greenhouse gas. The IPCC Guidelines for National Greenhouse Gas Inventories states:
"CO_2 emissions from biomass used as fuels are excluded from the total CO_2 emissions figure. The restriction of the Reference Approach[2] to fossil fuels results from the sustainable nature of biofuels. The CO_2 emissions are, however, reported for information purposes. Note that non-CO_2 emissions from biofuels are included in their respective inventories."

There are two accounting conventions that can then be used to handle the differences between renewable fuels and fossil fuels. One accounting convention is not to differentiate CO_2 emissions during the combustion process, and to treat the growth of the renewable fuel as a sink of greenhouse gases. Such sinks are mathematically represented as **negative** emissions. Hence, to work out the greenhouse gases emitted from the use of a renewable fuel, one needs to consider the whole life cycle of the fuel – from agricultural activity through to combustion. The terms 'exbodied' greenhouse gas emissions or full fuel-cycle emissions are also used for such life cycle emissions.

To avoid having to deal with negative emissions, another accounting convention has been developed. This second accounting convention, which is the one that we follow, treats fossil CO_2 differently from renewable CO_2 during the fuel combustion process. The exbodied greenhouse gas emissions will be the same whichever accounting convention is chosen.

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[2] The Reference Approach is the name given to one of the methods of greenhouse gas inventory quantification.

5 THE BIOFUELS INDUSTRY

5.1 Ethanol

Ethanol is currently produced in Australia from by-products such as wheat starch and C molasses. There are three producers of 'industrial' ethanol (as distinct from potable grades of ethanol) in Australia (Bullock, 2002), which include:

- CSR distilleries at Sarina, Queensland and Yarravile, Victoria;
- Manildra Group at Bomaderry (near Nowra), New South Wales; and
- Heck Group at Woongoolba, Queensland.

With such a limited number of producers in the ethanol industry it is difficult to obtain accurate estimates of current production levels (for reasons of commercial confidentiality). Naughten (2001) estimated that approximately 115 ML of ethanol was produced in Australia in 1999-2000, of which 37 ML was fuel grade ethanol. This included:

- 50 ML produced from waste starch feedstock, 70% or 35 ML of which was directed to the transport fuel market; and
- 65 ML produced from C molasses, of which only 2 ML was directed to the transport fuel market (29 ML was sold locally and 34 ML exported).

However, even in publishing these estimates, Naughten noted that the production of fuel ethanol had, reportedly, actually increased to a total of 53 ML in 1999-2000 (Naughten, 2001, p. 45). Bullock (2002) stated that the Manildra Group's ethanol production capacity, using waste and low-grade starch, was 50 ML. However, Bullock also indicated that the capacity at Manildra had been increased by around 50 ML (to 100 ML in total) with the installation of grains processing equipment (p. 6). In 2003 Environment Australia reported that ethanol sales from the Manildra facility had increased to 66.5 ML in 2001-2002 (which is consistent with Bullock's assessment that capacity at Manildra had been increased). Of this, it is believed that approximately 40 to 45 ML was fuel grade ethanol. Bullock also noted that a small batch fermentary has been installed in Woongoolba, Queensland that could produce 3 to 3.5 ML from C molasses feedstock. However, Bullock went on to say that competition for C molasses as a livestock feed means sourcing additional supplies of fermentables to the Woongoolba facility in future will be a significant problem.

The best estimate of the research partners involved in this study is that current fuel ethanol production in Australia is around 50 ML a year (Table 1). Of this, 45–48 ML is produced from waste starch and possibly degraded wheat feedstock (from the Manildra facility near Nowra), and 2–5 ML from C molasses (from the CSR, Sarina/Yarraville facilities and the Heck Group, Woongoolba facility).

How much further fuel ethanol production is likely to grow under current policy conditions is unclear. The economic viability of biofuel production in Australia is assessed in more detail in Chapter 7. The viability of ethanol production is heavily influenced by the availability and price of suitable feedstocks. Feedstock prices are typically volatile and strongly influenced by international market prices. For example, the most recent drought in Australia greatly increased the market price of grains, particularly sorghum and feed quality wheat, both of which approached $300/tonne in 2002-2003 (see Figure 5 in Chapter 7). Other recent studies that examine ethanol pricing in Australia have been undertaken by Naughten (2001) and Enecon (2002). CEC (1999) and Shapouri *et al.* (2002) also provide comparable information for the United States.

Table 1. Estimated production of fuel grade biofuels (ML)

	2002-2003 (best estimate)	Estimated 2009-2010 (reference case)
Ethanol		
Waste starch	45–48	80
C molasses	2–5	4–6
Sorghum/Feedwheat	0 [a]	0
Biodiesel		
Waste oil	0–2	30
Canola seed	0 [b]	0

[a] Ethanol production using degraded or feed quality wheat at the Manildra facilities is included with the estimated 50 ML of ethanol produced from waste starch.

[b] Environment Australia, in 2003, estimated that the production of biodiesel using canola feedstock was previously as high as 23 ML a year. Production has since declined markedly and reportedly is now only used to supply small volume vehicle trials.

The Sustainable Energy Development Authority of New South Wales (SEDA) suggested that all financially viable waste starch residues in New South Wales are already fully utilised, which is consistent with the total capacity of the Manildra facility being approximately 50 ML using waste and low-grade starch and 50 ML using grains (SEDA, personal communication, October 2003). An ethanol production plant planned for the Mossman Central Mill in northern Queensland would also provide a significant increase in production capacity (30 ML). This plant proposed to use C molasses supplies from the region initially, but then step up the level of production using sweet sorghum in later years. Mossman Central Mill and the Douglas Shire Council were jointly awarded up to $7.35m of funding from the Australian Government for a range of greenhouse gas abatement activities, including this ethanol plant. We understand development of the ethanol plant and deployment of the GGAP funds are currently on hold in Mossman.

In addition to the Mossman GGAP project, up to $8.8m is available to support a renewable fuels program for the east coast of Australia. BP's Bulwer Island Refinery on the Brisbane river will be installing appropriate infrastructure for the storage, blending and delivery of ethanol based fuels (i.e. E10) within the supply region of the refinery. Importantly, it should be noted that to access GGAP funding, proponents do need to achieve agreed greenhouse abatement milestone targets. However, given the results of the viability analysis presented in Chapter 7, and assuming no change in current government policy (particularly with respect to the phasing out of current ethanol production grants), it is not clear that these or other current plans to increase ethanol production capacity in Australia will come to fruition.

For the purposes of assessing the mix of fuels, which would go to make up a 350 ML biofuels target, it has been assumed that fuel ethanol production could be as high as 85 ML in 2010.

5.2 Biodiesel

Current production of biodiesel in Australia is relatively minor with a number of operations producing small volumes, although it appears output is likely to increase in 2003-2004 (Table 1). According to a study undertaken for Environment Australia, as at mid-2002 there were three commercial producers of biodiesel in Australia with a combined output of around 23 ML/year, most of which was from one plant (using canola oil feedstock). In 2003 Biodiesel Industries Australia also commenced production from a plant at Rutherford (NSW) that has a capacity of 14-17 ML per year based on using waste cooking oil feedstock. The bulk of current Australian production is being sold as BD100 in a small number of independent service stations, though Newcastle City Council has been running vehicles using BD20 for over a year (see http://www.ncc.nsw.gov.au/services/environment/ameif/whatsnew.cfm). Biodiesel plants are located in Victoria, NSW and Tasmania.

The biodiesel production identified by Environment Australia was based primarily on using canola oil feedstock, although tallow and waste oils (vegetable and fish) were also used in small volumes. However, due to increases in the price of canola associated with the drought in 2001 to 2003, production levels are thought to have been reduced significantly (from the 23 ML/year level estimated by Environment Australia) so that current production levels from this source are thought to be sufficient to only supply enough biodiesel for small volume vehicle trials. It is our assessment that without the continuation of current production grants, biodiesel production based on canola feedstock (whether whole grains or canola oil) is not viable over the longer term (see Chapter 7).

Small volumes of biodiesel are also produced from waste cooking oil feedstock. A new processing facility was recently opened in March 2003 (the Biodiesel Industries Australia plant at Rutherford, near Newcastle, New South Wales) with a capacity of around 14–17 ML a year which uses recycled cooking oils and processed vegetable oil feedstock, much of which will probably be collected from nearby McDonald’s outlets. The production of biodiesel in 2003-04 is expected to increase to around 10 ML.

Over the medium term, biodiesel production using canola feedstock is expected to remain economically marginal, at best (as discussed in more detail in Chapter 7). In contrast, biodiesel production using recycled cooking oil is on the cusp of being economically viable, even in the absence of direct government assistance. Certainly, if the Rutherford plant proves to be successful there would appear to be significant potential for other facilities to exploit the commercial synergies between the production and distribution of cooking oils, the food industry and the production of biodiesel. Over the medium term, the annual production of biodiesel in Australia is expected to increase, albeit modestly, to around 30 ML by 2010.

6 GOVERNMENT POLICY CONTEXT

Over the period to 2012, there are planned changes to transport fuel excise arrangements and to vehicle emissions and fuel quality standards (in some cases, already legislated and being implemented) that will affect the viability of and demand for the various transport fuels, as well as the net environmental impact of any increased biofuels use. In addition, there are measures relating specifically to the biofuels industry and biofuels use. This chapter outlines the principal policy measures.

6.1 Fuel Excise and Assistance Arrangements

As part of the 2003–04 Budget, the Australian Government announced reform of fuel tax arrangements to bring all currently untaxed fuels into the excise and duty system from 1 July 2008 (Australia.Treasury 2003, pp. 40–41). The aim of the reforms is to establish a broad sustainable taxation framework for fuels, by addressing anomalies in the current fuels tax system and providing increased long term certainty for investors. Fuels that will become excisable from 1 July 2008 are to include liquefied petroleum gas (LPG), liquefied natural gas (LNG) and compressed natural gas (CNG), where these fuels are used in internal combustion engines.

Biodiesel became subject to excise (and customs) duty from 18 September 2003 at the same rate as petrol and diesel. However, the effective excise payable on biodiesel is zero until 1 July 2008, since the Government proposes to offset excise with grants to importers and producers of biodiesel as a final fuel (and final fuel blends containing biodiesel). Similarly, in the period prior to 1 July 2008, the effective excise payable on ethanol is zero.

The final excise rates to be applied to LPG, ethanol and natural gas were announced on 16 December 2003. A copy of the media release, including a detailed outline of the excise rates to be applied, is included in Appendix V. This announcement was made after the analysis in this study was completed.

The effective excise rates assumed to apply to ethanol and biodiesel between 1 July 2008 and 1 July 2012 that were used in this study are outlined in Table 2.

Table 2. Estimated effective excise rates between 1 July 2008 and 1 July 2012 (cents per litre)

Fuel	2007	2008	2009	2010	2011	2012
Ethanol	0	5.22	10.44	15.66	20.88	26.1
Biodiesel	0	7.629	15.257	22.886	30.514	38.143

Source BTRE estimates.

In order to encourage new entrants to the biofuels industry, the Government also announced on 25 July 2003 that it will provide up to $37.6 million to fund a capital subsidy for projects that provide new or expanded biofuels capacity. The subsidy will be provided at a rate of 16c/L of additional capacity to viable projects producing a minimum of 5 million litres of biofuel and will be limited to a maximum of $10 million per project[3]. Biofuel proponents are required to

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[3] In the context of the 2001 election, the Government put forward a subsidy of 16c/L of new or expanded biofuel capacity, with a view to implementing an objective of having fuel ethanol and biodiesel contributing 350 million litres to the total fuel supply by 2010 (Liberal Party of Australia, 2001).

demonstrate that their project will be commercially viable in a post-excise relief environment and provide evidence of the prospect of firm contracts for the supply of biofuels for use in the domestic transport industry. Applications for the subsidy close during January 2004.

6.1.1 On and Off-Road Excise Rebates

Under the Energy Grant (Credits) Scheme (EGCS), businesses and individuals who use certain fuels in defined activities are eligible for relief of excise incurred in using these fuels. There are two credits available under the EGCS: an off-road credit and an on-road credit.

The off-road credit provides full excise relief for diesel fuel (and 'like' fuels) used in certain off-road activities in primary production, mining operations, rail and marine transport, hospitals, nursing homes and electricity generation at hospitality/retail and residential premises. Its primary objective is to maintain the competitiveness of the mining and primary production sectors. Since 2000, the scheme has also had the objective of reducing the diesel-fuel cost of rail and marine transport.

The on-road credit provides partial excise relief for diesel fuel used in on-road vehicles over 4.5 tonnes gross vehicle mass (GVM) that operate outside of defined metropolitan areas. Its purpose is to reduce transport costs and address urban air-pollution concerns. The on-road credit is also intended to maintain the pre-GST price relativities between diesel and alternative fuels by providing a credit for the use of certain alternative fuels (such as liquefied natural gas (LNG), compressed natural gas (CNG), ethanol and biodiesel) in activities that are eligible for a diesel on-road credit.

Grant rates under the EGCS are listed in Table 3. On 16 December 2003 the Prime Minister announced that details of further reforms are being developed and would be announced in 2004.

The combined impact of these possible excise and production assistance arrangements is shown in Figure 2, which illustrates the effective excise (i.e. excise net of production subsidies) for the different transport fuels to 2014–15. The excise on petrol and diesel fuels will be around 38c/L, with additional excise for high sulfur content fuels. Excise on LPG and NG will be introduced in 2008, rising to an assumed 29c/L (based on energy content) and 38.143c/L, respectively, by 2012. The effective excise on ethanol and biodiesel will also increase from zero in 2008 to an assumed 26.1c/L (based on energy content) and 38.143c/L, respectively, in 2012. We emphasise that these are assumptions about expected excise rates rather than calculations based on actual excise rates.

Table 3. Rates for The Energy Grants (Credits) Scheme[a]

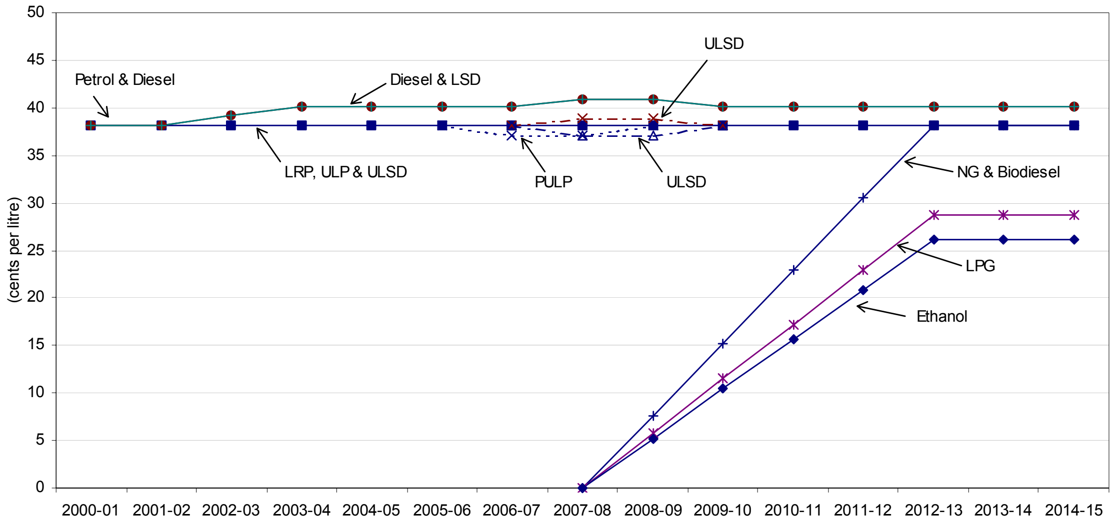
Activity	Product	Rate
Road transport	Diesel	$0.1851 per litre
	Liquefied petroleum gas (LPG)	$0.11925 per litre
	Ethanol	$0.20809 per litre
	Compressed natural gas (CNG)	$0.12617 per cubic metre
	Liquefied natural gas (LNG)	$0.0813 per litre
	Bio-diesel	$0.1851 per litre
Agriculture	Diesel and 'like' fuels[b] at the diesel rate	$0.38857 per litre (1/7/03-31/12/03)
		$0.39286 per litre (1/1/04-30/6/04)
		$0.39000 per litre (1/7/04-31/12/04)
		$0.38714 per litre (1/1/05-30/6/05)
		$0.38429 per litre (1/7/05-31/12/05)
		$0.38143 per litre (from 1/1/06)
	Like fuels[b] at the burner rate	$0.07557 per litre
Specified industrial uses	Diesel and 'like' fuels at the diesel rate	$0.38143 per litre
	Like fuels[b] at the burner rate	$0.07557 per litre
Use in burners	Specified diesel	$0.30586 per litre
All other eligible activities	Diesel (including marine diesel)	$0.38143 per litre
	Like fuels[b]	$0.07557 per litre

[a] Rates effective from 1 July 2003.

[b] Like fuels include: (i) heavy fuel oil, (ii) light fuel oil; and (iii) all fuels that attract the same rate of duty as diesel (except gasoline, coal tar, and coke oven distillates).

Source ATO (2003), Rates for the Energy Grants (Credits) Scheme (http://www.ato.gov.au/).

Figure 2. Possible effective excise on transport fuels for the period 2000–2001 to 2014–2015



Note Leaded petrol not included as it is no longer available under the fuel quality standards. Excise rates do not reflect the Government's announcement of 16 December 2003.
Sources BTRE estimates.

6.2 Fuel Quality and Efficiency Measures

The *Measures for a Better Environment* package, announced by the Prime Minister in May 1999 as part of *A New Tax System*, included a range of measures to improve air quality. Motor vehicle related measures included announcement of introduction dates for Euro II, III and IV (diesel) vehicle emissions standards and complementary fuel quality measures.

6.2.1 Fuel Standards

National standards for fuel quality are set and enforced under the *Fuel Quality Standards Act 2000*. The fuel standards are designed to reduce the level of pollutants and emissions from the use of fuel, improve vehicle operation, and facilitate the introduction of the new vehicle emission control technologies required to achieve the tighter emission standards. The first suite of national fuel standards came into force on 1 January 2002 and additional standards have been introduced since that time.

The *Fuel Standard (Petrol) Determination 2001,* and the *Fuel Standard (Automotive Diesel) Determination 2001* specify allowable limits on the composition and characteristics of petrol and automotive diesel fuels. Sulfur content is regulated for both petrol and diesel. The *Fuel Standard (Petrol) Determination 2001* also includes standards on the relative content of benzene, olefins, aromatics and the level of oxygen present in petrol. The petrol standards effectively banned the use of lead in petrol from 1 January 2002.

In addition, the *Fuel Quality Standards Act 2000* governs the use of alternative fuels - both as additives and as fuels in their own right. A 10% ethanol limit in petrol came into force in July 2003 and a biodiesel standard came into force on 18 September 2003. A standard for Automotive Liquefied Petroleum Gas is expected to take effect from March 2004 and standards for other fuels, including ethanol, are under consideration. (Appendix IV provides details of the Australian fuel quality standards for petrol, diesel and biodiesel.)

The fuel standards for petrol and diesel - most notably sulfur levels - become progressively tighter over the period to 2006, and this trend is expected to continue in the post-2006 standards. On 1 January 2005, a limit of 150 ppm sulfur will apply for all grades of petrol; and on 1 January 2006, sulfur in diesel will be limited to 50 ppm. Future fuel standards are under development and are likely to further reduce the sulfur levels of both petrol and diesel. The current proposed scenario includes: 50 ppm sulfur PULP by 1 January 2008; 10 ppm sulfur PULP by 1 January 2010; and 10 ppm diesel by 1 January 2009.

6.2.2 Fuel Quality Production Incentives

The fuel quality measures announced in the *Measures for a Better Environment* package included an increase in the excise on diesel with sulfur content above 50 parts per million (ppm) of 1c/L from 1 July 2003 and a further 1c/L from 1 January 2004. From 1 July 2003, the Government began paying an increment on the regular grant rate for eligible agricultural activities, to partly offset the rise in excise on diesel with sulfur content above 50 ppm. From 1 January 2007 the Government will increase excise (and customs) duty on all diesel to fund grant payments by the amount required to fund grant payments for the production or import of diesel with less than 10 ppm sulfur.

Also as part of the 2003–2004 Budget, the Government announced additional measures to encourage the production and use of cleaner fuels. From 1 January 2006, the Government will increase excise (and customs) duty on petrol for a period of two years by the amount required to fund grant payments for the production or import of premium unleaded petrol with less than 50 ppm sulfur (Australia. Treasury 2003, p. 224). From 1 January 2007, the Government will increase excise on diesel with a sulfur content of greater than 10 ppm. The indicative excise duty increase would be 0.7c/L on all diesel and 0.06c/L on all petrol, which would be used to fund subsidies for the increased production costs of around 1.1c/L for 50 ppm sulfur premium unleaded petrol and 1.0c/L for 10 ppm sulfur diesel.

It is anticipated that these measures will bring forward production of higher quality fuels prior to the date at which they will become mandatory under the provisions included in the *Fuel Quality Standards Act 2000*. The Government flagged that this initiative will be reviewed in the period prior to implementation to ensure that it aligns with the timing of new fuel standards and market conditions.

6.2.3 National Average Fuel Consumption Targets

The Australian automotive industry has committed to improving the fuel efficiency of new passenger motor vehicles by 18% by 2010, to 6.8 litres per 100 kilometres travelled (Kemp and Macfarlane, 2003). Further improvements in National Average Fuel Consumption (NAFC) are envisaged beyond 2010.

Meeting the NAFC target will require improvements in both vehicles and fuel quality. Reducing the sulfur content of fuels is a critical fuel quality parameter to enable advanced engine technology for improvement of emissions and fuel efficiency (Coffey Geosciences, 2003, p. 50). Sulfur reduces the efficiency of catalysts and interferes with the on-board diagnostic equipment.

6.2.3.1 Octane Enhancement

Enhancing the octane rating of petrol will also be critical in meeting the NAFC target by 2010, and may provide an opportunity for increased use of ethanol as a transport fuel. Issues associated with the use of ethanol to enhance octane are outlined in Chapter 11.

Methyl tertiary butyl ether (MTBE) is currently the most widely used fuel oxygenate and octane enhancer in Australia. MTBE will be effectively banned from 1 January 2004, due to its high potential to contaminate groundwater.

6.3 Vehicle Emissions Standards

The introduction of tighter fuel standards is linked to the introduction of more stringent vehicle emissions standards. Australian vehicle emissions standards are controlled through the Australian Design Rules (ADR). ADRs 37/00, 79/00 and 79/01, *Emission Control for Light Vehicles*, mandate standards for the emissions performance of new passenger vehicles and ADRs 70/00, 80/00 and 80/01, *Emission Control for Diesel Vehicles*, prescribe standards for emissions from new diesel powered vehicles. Australian vehicle emissions standards essentially involve the gradual adoption of European vehicle emissions standards (UN ECE vehicle emissions standards, generally referred to as ‘Euro’ standards). Many other countries, including Japan, Thailand, India and China, are also moving to bring their vehicle emissions standards into line with European standards (MVEC, 2003).

The timing for the introduction of new vehicle emissions standard for petrol fuelled vehicles is:

- Euro II in 2003–2004 for all new petrol vehicles; and
- Euro III in 2005–2006 for all new petrol vehicles.

The corresponding timetable for diesel fuelled vehicles is:

- Euro II in 2002–2003 for all new diesel vehicles;
- Euro III in 2002–2003 for all new medium and heavy-duty diesel vehicles; and
- Euro IV in 2006–2007 for all new diesel vehicles.

Appendix IV also outlines the emissions limits controlled under the Euro II, III, IV and V (diesel) vehicle emissions standards, for both petrol and diesel fuelled vehicles.

6.3.1 Future Vehicle Emissions Standards

European vehicle emissions standards also encompass the introduction of tighter emissions standards for gasoline fuelled vehicles from 2005 (Euro IV automotive gasoline vehicle emissions standards) and lower NO_x emissions from heavy-duty diesel vehicles from 2008 (proposed Euro V diesel vehicle emissions standards).

The Motor Vehicle Environment Committee has commenced the process for reviewing the appropriateness of introducing these standards in Australia (MVEC, 2003). While there is, as yet, no decision as to whether or when these standards might be introduced, it is highly likely that they will be introduced before the end of 2010, within the time frame of interest for this study. This would also be consistent with the already announced fuel quality production incentives (refer Section 1.2.2). The study has assumed that Euro IV (petrol) vehicle standards will be mandated in Australia from 2008 and Euro V (diesel) vehicle standards will be mandated from 2009.

6.3.1.1 Ethanol and Biodiesel Excise Arrangements

Up until 18 September 2002, ethanol used in transport fuels was excise free. In 2002, the Australian Government announced that from 18 September 2002, all ethanol used in transport fuels would be excisable at a rate of 38.143c/L, and that it would provide a producer subsidy, for ethanol produced in Australia, as a targeted means of maintaining the use of biofuels in transport (Australia. Department of Finance and Administration, 2002). Ethanol sourced from biomass feedstock for use in transport fuel was to be subsidised at a rate of 38.143c/L for eligible ethanol. The interim arrangements were to remain in place for 12 months from 18 September 2002 to 17 September 2003.

In the 2003–2004 Budget (Australia. Treasury 2003, p. 223), the Government announced that from 1 July 2008, it would provide grants to the producers and importers of fuels that are currently exempt from excise duty and which are used in internal combustion engines. The grants are to be progressively reduced, raising the effective excise (that is, excise less grant) for untaxed fuels from zero prior to 1 July 2008, to their final rates in five even annual steps commencing from 1 July 2008.

In addition, the Government announced that it would continue to provide production grants for fuel ethanol from 18 September 2003 on the same basis as announced in the *Mid-Year Economic and Fiscal Outlook 2002-03*. The Government announced that it would also provide grants for the production and importation of biodiesel, from 18 September 2003. Both grants are to be reduced in five equal annual instalments from 1 July 2008.

7 INDUSTRY VIABILITY

The aim in this chapter is to assess the viability of the Australian biofuels industry in comparison with traditional transport fuels (petrol or diesel).

7.1 Background

In this report, industry viability is assessed on the basis of whether or not biofuels production is considered likely to be cost competitive with traditional fuels over the medium to longer term; that is, after the period 2008 to 2012 when current government assistance arrangements are set to be phased out.

In this context the cost of producing biofuels relative to petrol and diesel is the single most important factor. Neither the availability of capital nor market potential is considered to be a barrier to the longer-term viability of an Australian biofuels industry. Issues related to the environmental performance of biofuels are considered in Chapter 10.

This analysis assumes current government policy in terms of capital subsidies, production subsidies and import tariffs for the biofuels industry will operate as currently proposed and that proposed changes in fuel standards in Australia, including the introduction of an ethanol standard, will proceed as currently planned.

The terms of reference refers to ‘biofuels in the Australian transport mix’. The two main biofuels relevant in this context are ethanol and biodiesel.

Production grants for fuel ethanol were introduced in 2002 to offset the effect of ethanol being brought into the excise system and currently fully offset the excise of 38.143c/L. These production grants will continue until 30 June 2008. The grants will be reduced to zero in 5 equal annual steps from 1 July 2008 to 1 July 2012. For biodiesel, excise and customs duty now applies at the same rate as the excise duty on ultra low sulfur diesel. As with ethanol, production grants will apply to fully offset the excise of 38.143c/L until 2008 and then will be reduced in five equal annual steps. As mentioned previously, new excise rates to apply from July 2008 were announced on 16 December 2003 and are outlined in Appendix V.

Given the relatively short lead time to 2008–2012 and the fact that this review — for reasons of tractability — is to draw on *existing* literature, studies and scientific reports, the analysis of industry viability focuses only on those biofuels that are currently technically viable. Technical viability refers to the case where all relevant technologies associated with production and use, suitable feedstocks, implications of waste disposal etc are available in a commercial context. That is, the analysis focuses on situations that do not require further extensive research and development or more than minor modifications to vehicle technology. In addition, while there is limited publicly available information on biofuel production options in Australia that are technically viable (referring specifically to cost information in this case), there is even less information available on prospective options. Further, speculation about how costs of production may decline in the future with the development of new technologies is just that, and does not provide a firm basis for decision making.

The production of ethanol from sugar using waste starch, molasses or whole grains (such as corn, sorghum or wheat) does not pose any technical problems, although there may be opportunities for cost savings in the future through refinements such as new enzymes. Similarly, the production of biodiesel from waste oil products (such as waste vegetable oil), tallow or whole grains (such as soybean, sunflower or canola) is commercially established in Europe and the United States.

In contrast, ethanol production from lignocellulosics is largely at the research and development stage, as is the new ZeaChem process, which it is hoped will reduce ethanol production costs in the longer term. While both options are potentially prospective, particularly when viewed over the longer term, the economics of each are not yet well established. In the case of the ZeaChem process, the probability that operations might be developed which are cost competitive with traditional transport fuels is very low at this point in time. For these reasons, both of these options (ethanol production from lignocellulosics or the ZeaChem process) have not been included in the assessment reported here.

Ethanol can be used as a fuel ‘extender’ where a blend of ethanol is added to petrol without adversely affecting performance or durability, hence extending or increasing the total volume of petrol (or the petrol-ethanol blend). Ethanol can also be used as a direct substitute for petrol or diesel in dedicated ethanol vehicles. Ventura Bus Lines in Victoria are currently running 3 buses on neat ethanol, the fuel being manufactured by CSR from C molasses, a by-product of sugar manufacture.

Ethanol is also an oxygenate and an octane enhancer (refer Section 8.1.2). MTBE (methyl tertiary butyl ether) is currently the most widely used fuel oxygenate, due to its combination of technical advantages and supply availability. However, MTBE will be banned in Australia from 1 January 2004 because of its high potential to contaminate groundwater. As well, fuel standards are becoming increasingly stringent in an effort to improve vehicle emissions and operation (refer Section 6.2). Both of these developments may represent a market opportunity for ethanol, both as a low sulfur fuel and as an octane enhancer. For example, demand for ethanol as an octane enhancer may increase as the demand for 95 RON petrol increases in line with greater up take of high octane vehicle technology, and as other octane enhancing properties of petrol (such as benzene, aromatic and olefin content) are reduced or capped under proposed new fuel standards in 2006. However, there is a range of alternative options to enhance octane levels and some literature suggests ethanol is one of the most, if not the most expensive option.

Ethanol can also be used in an emulsified mixture with diesel, known as diesohol. There are no present plans for widespread or large-scale use of diesohol in Australia.

In this analysis the viability of the ethanol industry is assessed on the basis of its competitiveness with petrol as an extender and not in its capacity as an octane enhancer.

Biodiesel can be used as a replacement or extender for diesel fuels. In the United States, biodiesel is commonly mixed with ordinary petroleum diesel to make biodiesel blends, such as BD5 (5% pure biodiesel and 95% petroleum diesel) or BD20 (20% pure biodiesel and 80% petroleum diesel). As with ethanol there are also technical advantages and disadvantages to the use of biodiesel, in terms of engine operation and performance. However, unlike ethanol, 100% biodiesel fuel can be used in existing diesel engines with only minor modification, although there are concerns about whether or not vehicle warranties may be voided.

For ethanol or biodiesel to be assessed as cost competitive, the cost of producing each needs to equate with (or ideally be lower than) the cost of those products for which they readily substitute, in equivalent energy terms - petrol in the case of ethanol and diesel in the case of biodiesel.

It is worth noting that numerous studies assessing the viability of ethanol as a transport fuel have been conducted in Australia over the past two decades. The conclusions in most of these earlier reports were not dissimilar to more recent reports. That is, ethanol is not economically viable without substantial levels of ongoing government assistance. Somewhat in contrast, fewer studies of the economic viability of biodiesel have been conducted in Australia. However, recent analysis concluded that it is possible that some new biodiesel projects using waste feedstocks could be viable. In this case, however, it was concluded the limited availability of low cost feedstocks would be likely to limit its potential future contribution to biofuels production in Australia. It was also concluded that biodiesel produced from dedicated energy crops (such as canola) is not economically viable in Australia without ongoing government assistance.

The remainder of this chapter is organised as follows. In the following section the relationship between world prices of crude oil and Australian petroleum product prices is discussed and ABARE's assessment of the medium term outlook for world crude oil prices is presented. Based on this benchmark, ex refinery prices for petrol and diesel and equivalent threshold prices for ethanol and biodiesel are estimated. Following this, the cost of producing ethanol and biodiesel is discussed in detail and longer term viability assessed.

7.2 Australian Petrol and Diesel Prices

By world standards, Australia is a small producer and consumer of crude oil and petroleum products, as well as importing and exporting crude oil and petroleum products. As a result, Australian producers and consumers of oil act as price takers in the global oil market. Reflecting this, changes in the price of petroleum products in Australia (ex refinery) mainly reflect changes in the world price of crude oil and exchange rates.

In recent years the price of crude oil has been at historically high levels; for much of 2003 oil prices have been above US$26 a barrel in world average trade weighted terms (around US$30 a barrel on a West Texas Intermediate basis). However, the price of crude oil fluctuates regularly reflecting the interaction of global business cycles and ad hoc global developments. ABARE's current forecast is for a gradual easing in oil prices over coming months and into 2004 (Haine *et al*., 2003). On a world average trade weighted basis, ABARE forecasts world crude oil prices to average US$27 a barrel in 2003 and US$24.75 a barrel in 2004. Over the medium term ABARE forecasts world oil prices to ease further to settle around the US$21 level in the period from 2008 – 2012 (in 2003 dollars). This is equivalent to around US$23 a barrel in West Texas Intermediate terms. This assessment largely reflects the potential for Iraq to increase output significantly over the medium to longer term and the availability of some unused OPEC capacity.

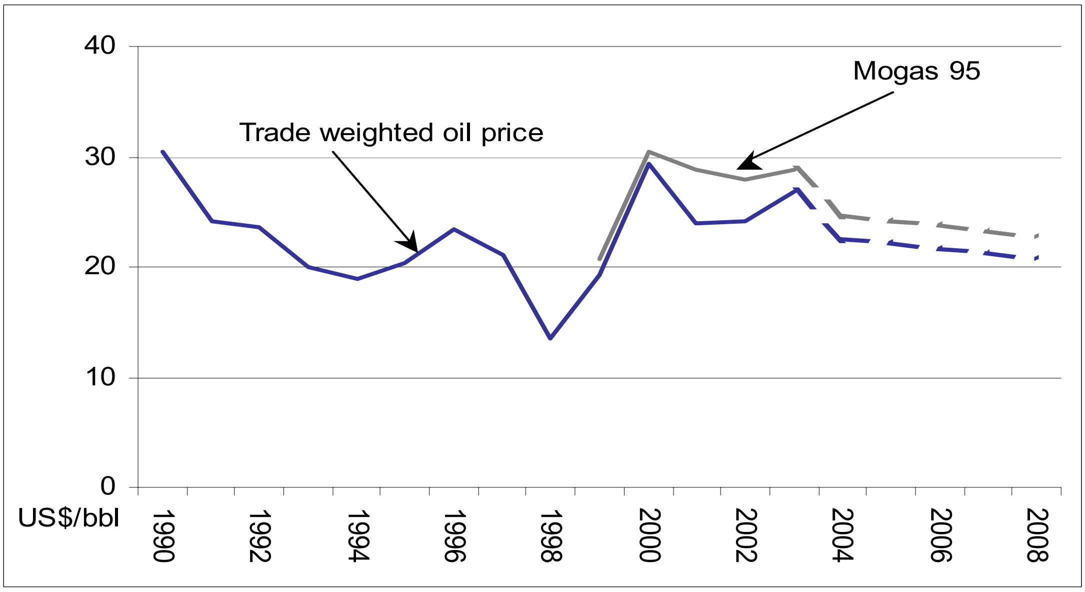
The current outlook for world oil prices presented by the International Energy Agency is also for world oil prices to ease to around US$21 a barrel in the period to 2010 before rising moderately thereafter to around US$25 a barrel by 2020 (IEA, 2002).

Most importantly for this assessment, the outlook for world oil markets is not for a sharp increase in prices, which might assist the competitiveness of alternative transport fuels, but rather for an easing in prices over the period to 2008–2010.

Singapore is the third largest refining and marketing centre in the world and is the closest major market to Australia. It is the most likely source of imported petroleum products (i.e. refined products) into Australia. Singapore's refineries are widely regarded as operating close to international best practice. Reflecting the open nature of the Australian economy, agencies that seek to benchmark Australian fuel prices, such as the Australian Competition and Consumer Commission and the West Australian Department of Consumer and Employment Protection, do so against movements in the prices of refined petroleum products sourced from Singapore.

In the case of unleaded petrol, Australian petrol prices are linked to the spot price of Singapore Mogas 95 Unleaded. In the case of diesel, Australian product prices are linked to a combination of the spot prices of Singapore Gasoil (80%) and Singapore Kerosene (20%) (ACCC, 2002). These refined product prices are also highly correlated with world prices. In particular, Mogas 95 usually trades within a US$3.10 band around the price of West Intermediate Index (Figure 3).

Figure 3. Real oil and petrol prices



The second factor influencing domestic product prices is the exchange rate. The Australian exchange rate depreciated significantly from 1996-97 through to 2001-2002 before recovering since. ABARE currently assumes the Australian dollar will average around US68c in 2003-2004 before returning to a trend level of US65c in the long run.

To determine gasoline prices in Australia, the usual approach is to use the Mogas 95 price, add a transport cost (approximately US1c/L) and convert this sum to Australian dollars. For this forward looking analysis the following approach, used by the ACCC, was adopted to create a Mogas 95 price series (ACCC, 2002). First, forecasts of the world average trade weighted price were converted to West Texas Intermediate terms by multiplying by 1.116. Second, $US3.1/bbl was added to cover refining costs.

For example, between 2008 and 2012 the world average trade weighted price of oil is expected to average around US$21 (in 2003 dollars). At an assumed exchange rate of US60c, the benchmark for an ex-refinery price of unleaded petrol would be A29c/L (in 2003 dollars). Similarly, the benchmark ex refinery price of diesel is estimated to be 33c/L (Table 5).

These estimates are relatively consistent with information provided independently that the cost of producing petrol and diesel (ex refinery) in Australia is 31 and 34c/L, respectively. In this the real cost of producing petrol and diesel in Australia, over the medium term, is assumed to average 29 and 33c/L, respectively.

As a point of comparison, if the world price of oil were to increase to US$30 a barrel (in 2003 dollars) by 2012 (rather than ease to US$21 a barrel), the benchmark ex-refinery price of unleaded petrol would increase to A40c/L (in 2003 dollars).

In comparing the estimated cost of ethanol and biodiesel with the prices of petrol and diesel it is important to ensure the cost terms are expressed in equivalent energy units. The energy content of biodiesel varies depending on the feedstock and the esterification process. Table 4 presents information on the calorific value of biodiesel produced from a variety of feedstocks and the energy density relative to diesel produced in Australia.

Table 4. Energy content of diesel and biodiesel

	Energy content MJ/L	Energy density relative to Australian diesel %
Diesel in Australia	38.60	1.00
Rapeseed methyl ester	34.04	0.88
Rapeseed ethyl ester	36.60	0.95
Canola methyl ester	34.71	0.90
Palm oil methyl ester	34.17	0.89
Tallow methyl ester	37.80	0.98
Soy methyl ester	36.08	0.93
Soy ethyl ester	38.10	0.99
Frying oil ethyl ester	36.28	0.94

For the remainder of this analysis, it is assumed that the relative energy density of biodiesel is 90%. The energy content (energy density) of a litre of fuel ethanol is 68% that of gasoline regardless of the feedstock. Accordingly, the benchmark prices for traditional transport fuels need to be scaled by the appropriate energy density to allow cost comparisons. In Table 5, the threshold ethanol and biodiesel prices are calculated, in equivalent energy terms and in real 2003-dollar terms. Without the current production grant or capital subsidy, the threshold price for ethanol is estimated to be 20c/L. That is, over the longer term (and in the absence of ongoing subsidy arrangements) only in those cases where the long run average cost of production (including an appropriate return on capital) is at or below 20c/L, is it likely that ethanol would compete with petrol and hence be assessed as viable.

Table 5. Threshold ethanol and biodiesel prices

		Ethanol		Biodiesel	
Estimated medium term ex refinery prices	c/L	29	(petrol)	33	(diesel)
Relative energy density	%	0.68		0.9	
Excise	c/L	38		38	
Capital subsidy[a]	c/L	1		1	
Threshold fuel prices					
– *including* excise relief and capital subsidy	c/L	59		69	
– *excluding* excise relief and capital subsidy	c/L	20		30	

[a] A one–off capital subsidy of 16c/L for a 40 ML plant amortised over the life of the plant reduces the fixed costs by approximately 1c/L.

In the case of biodiesel, the medium term threshold price in the absence of the production grant and capital subsidy is estimated to be 30c/L. Only in those cases where the long run average cost of biodiesel production is at or below 30c/L is it likely that biodiesel would compete with diesel and hence be assessed as viable.

Increases in fuel quality standards (for both petrol and diesel) and the need to find a suitable octane enhancer to replace MTBE, could see wholesale prices rise. Two reports by Coffey Geosciences (2000 and 2003) provide estimates of the impact on production costs of tighter fuel standards.

Coffey Geosciences (2000) estimated that the additional production costs of moving to Euro III (petrol) and Euro IV (diesel) fuel standards could be in the order of 1.1c/L for petrol and 1.5c/L for diesel. Coffey Geosciences (2003) also estimated that the additional production costs of moving from Euro III to Euro IV (petrol) and from Euro IV to Euro V (diesel) could be up to 4.3c/L for petrol and 0.7c/L for diesel. Hence, moving from current fuel standards to Euro IV (petrol) and Euro V (diesel) fuel quality standards could add up to 5c/L for petrol and 2c/L for diesel. Note, these estimates are expressed in terms of litres of petrol and diesel, hence 5c/L (petrol) equates to 3.25c/L (ethanol) in energy equivalent terms. Bearing this in mind, the relevant medium term threshold price for ethanol would increase to approximately 24c/L (in 2003 dollar terms) in the absence of any industry assistance, and the medium term threshold price for biodiesel would increase to around 32c/L (in 2003 dollar terms).

7.3 Biofuel Production Costs

The costs of producing ethanol and biodiesel using different feedstocks are discussed in detail in this section. Given the high degree of uncertainty that surrounds individual cost items, it is appropriate to work with ranges of costs. In the following discussion a low and a high range of cost estimates are provided where appropriate. While it is possible to estimate a range for most cost components, it is difficult to ascribe a probability distribution to these. Where there is evidence that costs may tend towards one end of the range, this is also highlighted in the discussion.

7.3.1 Fixed Capital Costs

The fixed capital costs and operating costs of ethanol and biodiesel production are similar across each of the various feedstock options. Estimates of these cost components are presented in Table 6.

Fixed capital costs reflect the rate of return on invested capital - whether that capital is equity based or debt financed. The required rate of return is assumed to vary between a low of 6% and a high rate of return of 8%. It is worth noting that in some project proposals, assumed rates of return of 20% (and higher) are not uncommon.

The low rate has been pegged to ABARE’s assumptions for the prime lending rate to large business over the medium term (Penm and Fisher, 2003). An alternative low rate is to consider the so-called ‘risk free’ rate, which is the return on ten-year government bonds, currently around 3% in real terms. However, this rate was considered too low for a business with the input cost and output price risks typically faced in the biofuels industry. The high rate of return is commensurate with long-term real returns from the Australian stock market.

For ethanol, Bullock (2001) suggests that, depending on the technology selection and the mode of operation, capital costs range from $40 to $60m for capacities ranging from 40 to 60 ML: or approximately $1m for each 1 ML of annual capacity. Across 11 currently proposed projects, total new capacity of 635 ML has been costed at approximately $620m, which again is broadly consistent with Bullock’s assessment. The average size of these proposals is 58 ML, ranging from 10 to 100 ML in capacity. Naughten (2002) suggested a 50 ML a year plant would cost approximately $68m.

With a rate of return of 6% and a plant life of 30 years, a 40 ML plant costing approximately $40m, represents a fixed capital cost requirement of approximately $2.9m a year or 7c/L. In the case where the real rate of return was assumed to be 8.0% a year, fixed annual capital costs would be 9c/L.

For biodiesel, 15 currently proposed projects with new capacity totalling 757 ML have been costed at approximately $597m. The typical proposal is structured around a 40 ML plant costing approximately $25m, although the proposals range in size from 5 to 100 ML in capacity.

With a rate of return of 6% and a plant life of 30 years, a 40 ML plant costing $25m equates to a fixed annual capital cost of around $1.8m or 4c/L. In the case where rates of return were 8% a year, this equates to a fixed annual capital cost of $2.2m or 5c/L. Across the range of current biodiesel proposals, from the smallest to the largest, capital costs (assuming a real interest rate of 6.0%) are estimated to vary between 4.3 and 8.6c/L.

Table 6. Fixed capital and operating cost estimates

	Operating Cost		Fixed capital costs	
	Low c/L	High c/L	Low c/L	High c/L
Ethanol	5	10	7	9
Biodiesel	5	10	4	5

If the plant life is considerably shorter than 30 years (as is often assumed to be the case in many project proposals), estimated capital costs would rise substantially. For example, with only a 10-year life, at a 6% rate of return, the capital costs for ethanol increase from 7c/L to 13c/L. For biodiesel, the capital costs would increase from 4c/L to 8c/L.

In considering the current policy of a one-off capital subsidy 16c/L for a qualifying plant built by 2006, then for a for a 40 ML plant, this subsidy is equivalent to reducing fixed costs of the plant by approximately 1c/L, assuming a 30-year plant life.

7.3.2 Operating Costs

Bullock (2001) suggests that for a 60 ML plant, ongoing costs excluding feedstock (i.e. labour costs, other input costs including energy, maintenance, depreciation, tax and financing charges) amount to 36c/L in total (p. 11). In that analysis Bullock assumed an internal rate of return (or hurdle rate) of 20% after tax. On this basis his estimate of the cost of capital (for a 60 ML ethanol plant) would be close to 20c/L and variable operating costs approximately 16c/L.

Based on ABARE analysis in deconstructing a number of currently proposed ethanol and biodiesel projects, this estimate of recurrent operating costs would appear to be an overestimate. It is ABARE’s assessment that operating costs (excluding the cost of feedstock and capital) for both ethanol and biodiesel production is likely to be in the range of 5 to 10c/L. These estimated costs are based on information taken from a variety of sources, although the detailed basis is commercially confidential.

7.3.3 Feedstock Costs and Additional Revenue

Feedstock costs account for the largest proportion of total costs of producing ethanol and biodiesel, hence the initial development of these industries in Australia has been focused around the supply of low cost waste products or by-products. Once these feedstocks are fully utilised the costs of production will tend to escalate quickly, as evidenced below. For example, using oil seeds or vegetable oil purchased from competitive markets rather than waste oil is likely to increase costs significantly. Similarly, in the case of ethanol produced from starch, dry milling whole grains (such as sorghum or feed wheat) as opposed to wet milling waste starch (a by-product of starch and gluten production) is also likely to increase costs significantly.

In some cases other revenue streams contribute to cover total costs. For example, in the case of both ethanol and biodiesel produced from whole grains, crushed grain meal (also called distiller's grain) is a valuable by-product for use as a livestock feed. Similarly, glycerin is a valuable by-product of the production of biodiesel. Revenue from associated by-products reduces the total revenue required to cover costs and hence is important to the overall calculation of industry viability.

7.4 Ethanol

Estimated feedstock costs and by-product revenue streams for ethanol production using different feedstocks are presented in Table 7. The options examined in more detail include the production of ethanol from:

- waste starch using existing capacity;
- C molasses using existing capacity;
- C molasses using new capacity;
- B molasses using new capacity;
- A molasses using new capacity; and
- degraded wheat or sorghum (i.e. feedgrains) using new capacity.

Table 7. Ethanol feedstock costs and by-product revenues

	Feedstock			**Meal revenue**			**Net required revenue**[a]
	yield	**price**	**cost**	**yield**	**price**	**revenue**	
	L/t	**$/t**	**c/L**	**kg/L**	**$/t**	**c/L**	
Existing capacity							
Waste starch	n/a	n/a	10 [b]	—	—	—	18
C molasses	280	50	18	—	—	—	26
New capacity							
Sorghum/Feedwheat	380	137	36	0.9	220	20	32
C molasses	280	50	18	—	—	—	33
B molasses	350	114	33	—	—	—	48
A molasses	450	250	56	—	—	—	71

[a] Assumes operating costs of 7.5c/L and capital costs of 8c/L. For this analysis, it has been assumed that plants do not qualify for the capital subsidy. Applying the subsidy would reduce the net revenue required by approximately 1c/L (depending on the size of the plant).

[b] ABARE estimate

Also included in Table 7 is the total net revenue required to break even. That is, total long run average costs (including a return to capital) less by-product revenues. It is this figure that is compared to the threshold price estimated earlier (20c/L) to determine whether or not the operation is likely to be viable over the medium to longer term.

7.4.1 Waste Starch

It is difficult to provide an accurate assessment of the total average cost of ethanol production using waste starch, principally because of the commercial sensitivity of this information. ABARE conservatively estimates that the cost of waste starch feedstock is unlikely to exceed 10c/L and may well be significantly lower. It is worth noting that the production of ethanol from waste starch was originally developed in 1992 as a waste disposal solution, enabling the production of higher quality starch and balancing out the effects of seasonal agricultural production. Hence the real cost of waste starch feedstock may in fact approach zero or be negative.

Despite this, however, there would appear to be limited opportunities to significantly increase the production of ethanol using waste starch resources beyond existing operations. It is the assessment of the Sustainable Energy Development Authority of New South Wales (SEDA[4]) that all financially viable waste starch residues in New South Wales are already being utilised to produce around 55-60 ML of ethanol a year. Hence, while production of ethanol from waste starch using existing facilities would appear to be a viable activity (the required revenue is significantly below the threshold price of 19c/L), there would appear to be limited potential to expand production capacity significantly. At the very least, the limited availability of waste starch feedstock would preclude the development of new production facilities.

7.4.2 C Molasses

The cost of the raw sugar by-product feedstock, C molasses, varies in a wide range depending on the supply of and demand for, exported raw sugar (which largely determines the domestic supply of C molasses) as well as the demand for molasses in other uses. This includes the production of ethanol required for industrial and potable purposes as well as for the stockfeed and food additive markets. In Figure 4 it can be seen that over the period from 1988 to 2002 the unit value of molasses exports has typically been lower than $100 a tonne and averaged $85 a tonne. What is not clear in these statistics (sourced from Australian Bureau of Statistics trade data) is the composition of the molasses trade. As the trade includes B and A molasses, the unit value of exports is likely to over estimate the prevailing market price of C molasses.

Naughten (2001) noted that the price of C molasses in Australia has typically ranged between $25–75 a tonne. In Table 7 ABARE has assumed the real price of C molasses will average $50 a tonne over the medium term.

The yield of ethanol from C molasses typically varies between 270–290 litres a tonne, depending on the sugar content of the feedstock. Taking the extremes of each of these ranges (cost per tonne and yield) the cost of C molasses feedstock is estimated to vary between 9c/L and 27c/L, with the mid–point being approximately 18c/L. On this basis (and taking into account operating costs of 8c/L, though not a return to capital which is assumed to be sunk) the long run average cost of ethanol production based on C molasses feedstock is estimated to be approximately 26c/L.

ABARE's current medium term outlook for the global sugar market is for prices to ease from their current high level of US8.2c/lb to settle around US6.3c/lb (in real terms) by 2007-2008. In this case $50 a tonne for C molasses may prove to be high with market outcomes closer to the

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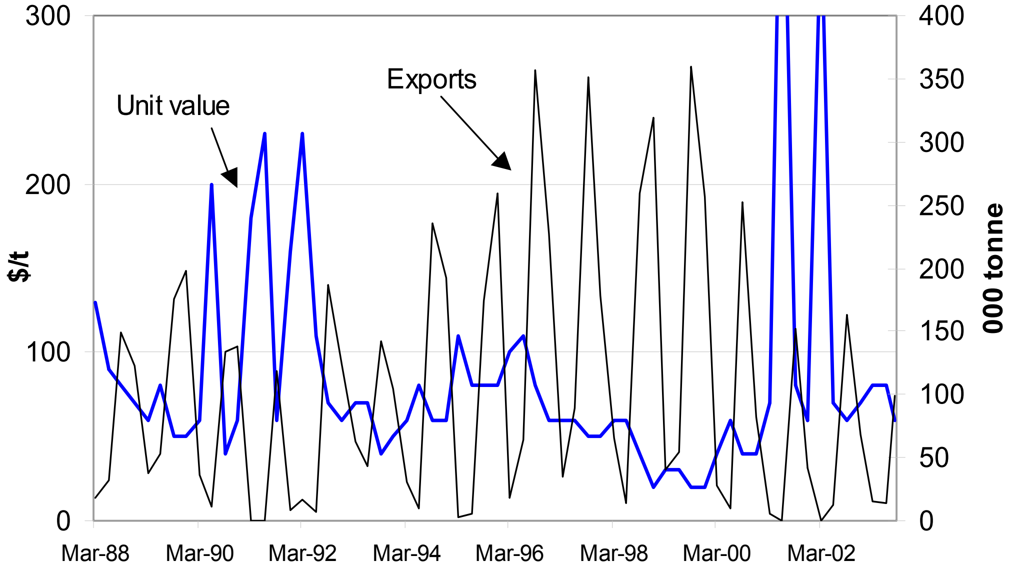
[4] SEDA is currently in the process of completing a handbook of the bioenergy industry in New South Wales which is expected to be published in 2004.

lower end of the price range noted above. Were the price of C molasses to be $25 a tonne, the cost of ethanol production would be reduced to only 16c/L.

Ethanol is produced either as hydrous or anhydrous ethanol. Hydrous ethanol contains approximately 5% water and is not miscible with petrol. Anhydrous ethanol contains greater than 99% ethanol and will readily blend with petrol. Most ethanol currently produced from C molasses in Australia is hydrated ethanol produced for industrial and potable markets (Naughten, 2001). To produce anhydrous ethanol, most ethanol plants use a molecular sieve to dehydrate hydrated ethanol and remove the last of the water. In estimating the cost of production from C molasses, using existing technology (Table 7), it is assumed that any additional investment in dehydration facilities that might be required will be completed and written down in the short term. Ongoing costs associated with dehydration are assumed to be included in recurrent operating costs.

The production of C molasses in Australia increased from around 600,000 tonnes in 1991 to 1.2 million tonnes in 1999. Domestic consumption of C molasses has remained relatively steady at 0.5–0.6 million tonnes over the past decade, with the residual exported (Figure 4). Only a small proportion of total C molasses production is currently used in fuel ethanol production in Australia (around 3–4 ML).

Figure 4. Molasses exports: quantity (kt) and value (AUD)



It has been suggested that a potential future case of interest is to assume all exports of C molasses could be diverted to fuel ethanol production (Bullock, 2001, p. 5). This resource would potentially provide the feedstock to supply around 180 ML of fuel ethanol. However, as noted by Naughten (2001) and as is evident in Figure 4, export volumes of C molasses follow a significant cyclical or seasonal pattern.

Taking only those quarters for which exports exceeded 100,000 tonnes since March 1988 (26 quarters out of 63 in total), the average unit value of molasses exports was $55 a tonne. However, for those quarters where trade was less than 100,000 tonnes, the average unit value of exports was $107 a tonne. In four quarters molasses exports were less than 400 tonnes with the unit value as high as $450 a tonne, which suggests that little C molasses was traded at all during these periods, with the bulk of this trade likely to have been B and/or A molasses.

The high variability of molasses supplies (both seasonal and year-on-year) presents a logistical problem to any large scale ethanol producers (or industry) who potentially face a relatively steady and growing demand from fuel blenders and ultimately vehicle drivers. To secure consistency of feedstock supply, ethanol producers may have to pay a premium above export parity prices. Alternatively ethanol producers will need to store feedstock (or final product) to accommodate the variability in feedstock supplies. In either case, the costs of production may be higher than indicated here.

Comparing the estimated net revenue required for ethanol production using C molasses (and using existing capital) with the threshold price of petrol, it can be seen that this option is only marginally viable at best and any increase in costs, such as those discussed above, would compromise this outcome. In the case where new capital expenditure is required (adding approximately 8c/L to overall costs), ethanol production using C molasses is estimated to cost 33c/L and is not viable. The cost of C molasses would need to fall to under $16 a tonne for this option to match the threshold price of 19c/L.

7.4.3 B and A Molasses

In principle, the production of sugarcane based fuel ethanol could be further expanded by making use of the A and B molasses currently used as precursors to raw sugar production. If all Australian mills were to divert their B molasses supplies to ethanol, then total sugarcane based ethanol production could be around 750 ML a year (Bullock, 2001). Diversions of A and B molasses between raw sugar and fuel ethanol uses are a feature of the Brazilian ethanol market (Naughten, 2001), although whether the Brazilian model is a realistic option in Australia's case is not clear. Naughten (2001) also quotes the Australian Biofuels Association as saying that 'the Brazilian model is not entertained in Australia' and 'the use of dedicated food crops such as sugarcane in Brazil, and corn in the United States currently have no place as [an] ethanol production model in Australia' (Naughten, 2001, p. 11).

As primary and secondary stages in the sugar production process, it is important to note that making use of B molasses means C molasses supplies are foregone, and similarly, using A molasses means B molasses supplies are foregone.

More importantly, as commodities that have valuable alternative uses, A and B molasses have relatively high opportunity costs. Bullock (2001) estimated that, depending on the price of C molasses and the final price of sugar, the cost of B molasses for ethanol production could range from 24 to 45c/L. In this analysis it has been assumed the cost of B molasses will average $114 a tonne over the medium term and the cost of A molasses $250 a tonne.

In both cases the yield of ethanol per tonne increases (to 350 litres a tonne for B molasses and 450 litres a tonne for A molasses), but in both cases, this increase in yield does not compensate for the increased cost. As a result, the feedstock costs for B and A molasses are both estimated to be significantly higher than for C molasses.

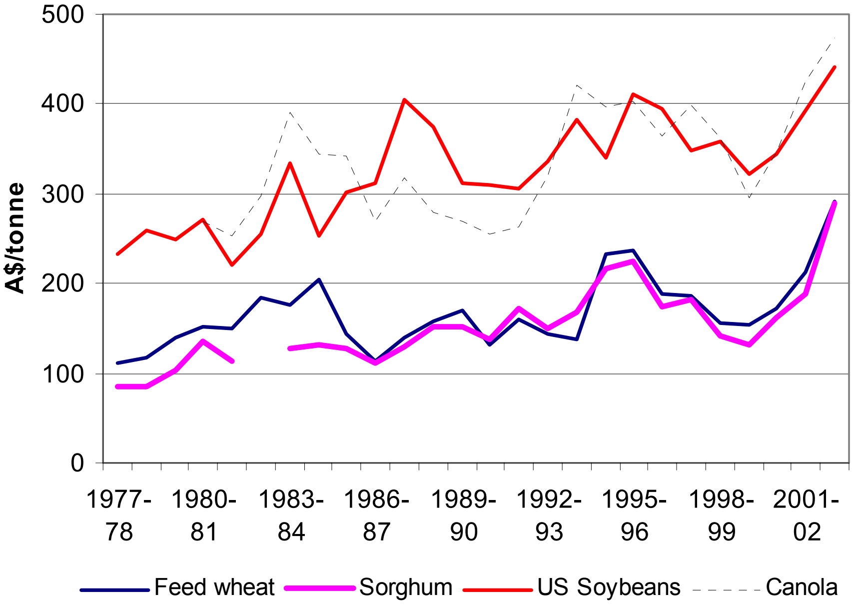
In total, the average cost of ethanol production using B and A molasses is estimated to be 48 and 71c/L, respectively. Compared with a threshold price of 19c/L, neither of these options are considered to be viable.

7.4.4 Whole Cereal Grains

While coarse grains and degraded or feed quality wheat appear to represent a tremendously large potential resource for ethanol production, production and prices vary significantly depending on seasonal conditions. ABARE estimates that approximately 3100 kt of wheat was supplied to the stockfeed market in 2001-2002, taking up around half of total domestic demand for feed grains (ABARE, 2003). However, anywhere up to 60% of the total wheat crop can be diverted to the feed market in any given year. Total Australian wheat production was over 24,850 kt in 2001-2002 but only 9385 kt in 2002-2003. During the most recent drought anecdotal reports suggest some livestock producers used prime hard wheat to feed livestock, illustrating the scarcity of supplies.

As illustrated in Figure 5, the price of feed wheat and sorghum have both stayed largely between $100 and $200 a tonne since the late 1970s, although the most recent impact of the drought is clearly evident, with the price of feed wheat and sorghum in 2002-2003 close to $290 a tonne. ABARE's medium term forecast for sorghum is for prices in real terms to ease from current high levels to average $137 a tonne in 2007-2008 (Connell *et al.*, 2003). On the basis that wheat and sorghum provide a yield of 380 litres a tonne, the estimated cost of these feedstocks is 36c/L.

Figure 5. Grain prices



As mentioned previously, one of the main co-products created in the production of ethanol from cereal grains is distiller's grain or grain meal. Distiller's grains are high in protein and other nutrients and are a highly valued livestock feed ingredient. With a yield of around 0.9 kg of meal per litre of ethanol produced and the cost of meal assumed to be $220 a tonne (on a dry basis), revenue from the sale of distiller's grain is estimated to amount to 20c/L, or almost 32% of the total average costs of ethanol produced from cereal grains.

Subtracting the meal revenue from total costs, the residual net revenue required (taking into account operating costs of 8c/L and capital costs of 8c/L) is estimated to be 32c/L. This does not compare very favourably with the threshold price of 20c/L.

In this case, the cost of feedstock and meal revenue are clearly the two largest items determining the residual net revenue required to break even. For the threshold price of 19c/L to be achieved, the cost of wheat or sorghum feedstock would need to fall to $96 a tonne (a price last achieved in 1978-79) or the price of distiller's grains would need to rise to over $460 a tonne (or some combination of both of these changes). It is ABARE's assessment that both of these outcomes are unlikely.

7.5 Biodiesel

Estimated feedstock costs and by-product revenue streams for biodiesel production using different feedstocks are presented in Table 8. The options examined in more detail include the production of biodiesel from:

- waste cooking oil using new capacity;
- tallow using new capacity; and
- whole grains or oilseeds (such as canola) using new capacity.

7.5.1 Waste Vegetable Oil

The biodiesel yield from all oil in liquid form (including waste cooking oil) is assumed to be 80%. Given the specific gravity of 0.92, a tonne of cooking oil yields 870 litres of biodiesel. SEDA estimates that feedstock costs for biodiesel vary from 20 to 90c/L. In this case it is assumed that the low end of SEDA's range refers to waste cooking oil while the upper end of the range is likely to refer to commercially grown oil seeds or vegetable oil. As with waste starch, it is difficult to determine the true economic value of waste cooking oil as no transparent market exists. Some businesses incur costs associated with the disposal of the waste cooking oil while others are paid for their waste oil. At this stage in Australia this is an undeveloped market. The Australian Tax Office estimated the price of waste cooking oil to be $170/tonne (Australian Tax Office, personal communication). On this basis the cost of waste cooking oil feedstock is estimated to be approximately 20c/L.

Table 8. Biodiesel feedstock costs and by-product revenues

	Feedstock			Chem–icals[bc]	Glycerol revenue[d]	Meal revenue			Net required revenue[e]
	yield[a]	price	cost			yield	price	revenue	
	L/t	$/t	c/L	c/L	c/L	kg/L	$/t	c/L	c/L
Waste oil	870[f]	170	20	9	6	—	—	—	35
Tallow	894[g]	450	50	9	6	—	—	—	66
Canola seed	370	353	95	9	6	24	140	36	76
Canola oil	875[h]	910	104	9	6	—	—	—	119

a. The yield of biodiesel per litre of oil is 0.8 litres. **b**. Methanol costs of $800/t at a specific density of 0.791 with 125ml/L of biodiesel required gives 8c/L input cost. **c**. Catalyst cost of $200/tonne at a ratio of 0.5% by weight equates to a 1c/L input cost. **d**. Glycerine yield of 8% per litre of biodiesel sold at $850/t with a specific density of 1.112. **e**. Assumes operating costs of 7.5c/L and capital costs of 4.5c/L. For this analysis, it has been assumed that plants do not qualify for the capital subsidy. Applying the subsidy would reduce the net revenue required by approximately 1c/L (depending on the size of the plant). **f**. At a specific density of 0.92. **g**. At a specific density of 0.895. **h**. At a specific density of 0.914.

One issue of critical importance is the costs associated with collecting waste cooking oil. Little information is available on this. Anecdotal evidence suggests waste cooking oil feedstock may be collected in conjunction with the delivery of new cooking oil; hence there may be commercial synergies between the production and distribution of cooking oils and the production of biodiesel. However, given the closeness of the comparison between the estimated cost of biodiesel production (35c/L) and the threshold diesel price (27c/L), it follows that collection costs will need to be minimised so as not to compromise the viability of this option. This suggests that biodiesel production from waste cooking oil is likely to be located close to densely populated urban areas. Facilities are likely to be relatively small outside of urban areas.

On the other hand, another important consideration is that a biodiesel industry in Australia would appear to be emerging. In this early development phase, it is possible that the ability to place a market value on waste cooking oil could lead to increased awareness (particularly within the food industry) and an increase in supplies as product is diverted away from current disposal methods. This could lead to a fall in the price of waste cooking oil over the medium term.

In this study it has not been possible to assess the relative impacts of neither increased costs associated with increased collection activity, nor potential reductions in waste cooking oil prices associated with increases in supply.

The cost of chemicals used in the production of biodiesel, mainly alcohol and catalyst, depends on the production process, as well as the prices of the chemicals. The continuous flow process requires the stoichiometric amount of chemicals (that is, the exact proportions required for the chemical reaction), whilst the batch process requires an excess of alcohol to drive the reaction to completion. However, in the batch process, most of the excess alcohol can be recovered (more than 90% of the excess can be recovered for use later) such that the differences in costs between the two processes are small enough to be ignored for this analysis. (McAloon *et al.*, 2000).

The amount of alcohol required for the reaction varies depending on the type and quality of the feedstock (in particular, the amount of free fatty acids in the oil) and the process. The amount required varies between 9 and 15% by volume. This analysis assumes that 125ml of methanol are required for every litre of biodiesel produced. Methanol is assumed to cost $800/t. This equates to approximately 8c/L of biodiesel produced.

The amount of catalyst required is assumed to be 0.5% by weight and is assumed to cost $200/t. This equates to approximately 1c/L of biodiesel produced.

Combining both the cost of alcohol and catalyst, the total cost of chemicals is assumed to be approximately 9 cents for each litre of biodiesel produced.

As mentioned previously, glycerin (or glycerol or glycerine) is a valuable by-product of the production of biodiesel. It is commonly used as a solvent, plasticiser and softening agent in a wide range of industries such as cosmetics, tanning and dying, food processing, chemicals and explosives. With a yield of 8% per litre of biodiesel produced and a price of around $850 a tonne, revenue from glycerol sales is estimated to be around 6c/L of biodiesel produced.

Taking all this together, the total cost of biodiesel production based on waste cooking oil feedstocks is estimated to be 35c/L. This compares with the threshold price for diesel estimated earlier of 27c/L. Given the high degree of uncertainty surrounding many of the cost items in this calculation, this difference is too small to conclude that biodiesel production based on waste cooking oil feedstocks is not viable in the medium to longer term.

This assessment then begs the question why a biodiesel industry based around utilising waste cooking oil feedstock has not already been developed in Australia? Until the introduction of biodiesel fuel standards (Appendix IV) there were no technical impediments to such a process or such an industry. The resulting biodiesel fuel could be used either as a replacement for diesel or as a diesel extender in vehicles that presently use diesel and has been used successfully in such applications in trials conducted in Europe. Information on the European experience is given in http://www.biodiesel.at/Userfiles/LiteraturPDF/Prossnigg-info-engl.pdf.

There is thus no simple answer to the question in the previous paragraph. It may be that the industry is now emerging and, over the longer term, production will increase significantly. Alternatively it may be the case that the industry will remain small for some time. However, what is clear is that of all the biofuels operations which require new investment, biodiesel production using waste cooking oil would appear to be the most competitively placed to compete with traditional transport fuels.

It is difficult to assess the quantity of waste cooking oil produced in Australia. On the basis that waste cooking oil is produced at a rate of between 10–12 litres per person (Australian Tax Office, personal communication), total Australian supplies would be between 220 and 260 ML in 2010 (assuming a population of 22 million). SEDA estimates that 120,000 tonnes of waste cooking oil is currently produced in New South Wales alone (personal communication). On the assumption that 50% of 264 ML of waste cooking oil[5] is recoverable (and assuming a yield of 80%), this resource could be used to produce between 90 and 105 ML of biodiesel.

7.5.2 Tallow

Tallow is rendered animal fat and a by-product of the livestock processing industry. Australian tallow production in 2000-2001 was approximately 567,000 tonnes (Australian Renderers Association, 2002), most of which was exported (68%). The biodiesel yield from tallow is approximately 894 litres a tonne. Since June 1994 tallow prices in Australia have largely been in the range from $400 to $650 a tonne (Figure 6). The unit value of exports averaged almost $510 a tonne over the period 1988-89 to 2002-2003. For this analysis it was assumed the real medium term price of tallow would average $450 a tonne (in real terms).

On this basis, and taking into account both fixed and recurrent operating costs as well as by-product revenue, the net revenue required to cover costs is estimated to be 66c/L. Compared with a threshold price of 27c/L this option is not considered to be viable.

7.5.3 Oil Seeds and Canola Oil

A considerable number of new project proposals are based on the utilisation of whole grain oilseeds, and canola in particular. However, as internationally traded agricultural commodities, oilseed prices vary considerably depending on both domestic market conditions (i.e. drought) and international market developments. ABARE's current medium term forecast for canola is for the price to ease considerably over the medium term from its current high level (see figure C) to average $353 a tonne (in real terms) in 2007-2008 (Connell *et al.,* 2003). As with ethanol produced from cereal grains, crushed grain meal is also a valuable co-product in the production of biodiesel from oil seeds. In this analysis it is assumed canola grain meal is priced at $140 a tonne providing a revenue credit of 36c/L of biodiesel produced.

Based on these figures the net revenue required to cover costs is estimated to be 76c/L, suggesting biodiesel production from canola is not viable in the medium to longer term without significant government assistance. In the case where the raw feedstock is canola oil rather than whole seeds the costs of production are even higher ($1.19 per litre) reflecting both the higher cost of the feedstock and the lack of a grain meal co-product.

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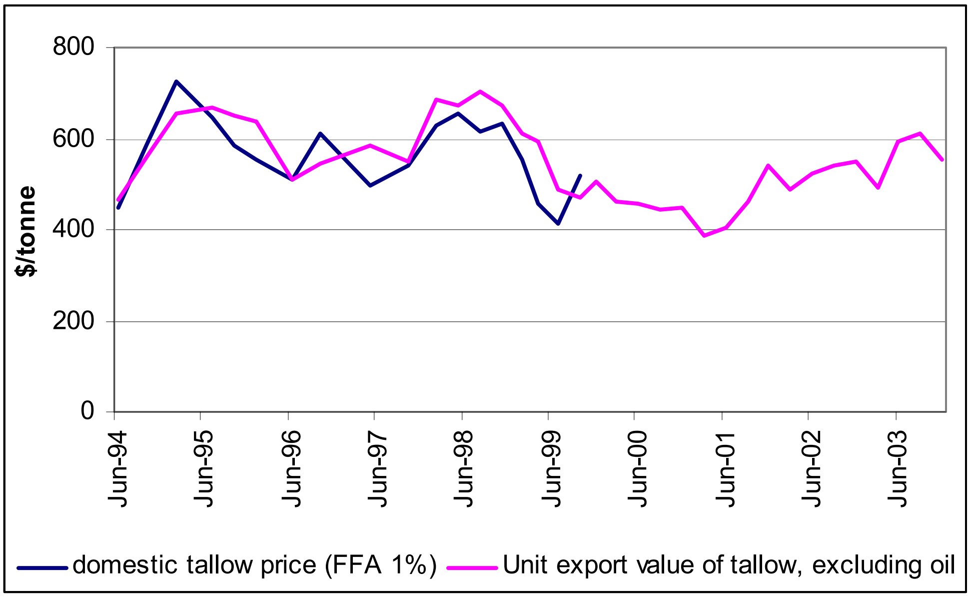
[5] http://www.biodiesel.org/resources/reportsdatabase/reports/gen/19970901_gen-190.pdf claims that in Austria 41% of waste oil is relatively easy to collect. A slightly higher figure would apply to Australia, which has a higher proportion of fast-food outlets.

Figure 6. Tallow prices



Source: Aginfo, Australian Bureau of statistics

8 EMISSIONS FROM ETHANOL USE

8.1 Petrohol (E10)

8.1.1 Background

Ethanol is probably the most widely used alternative automotive fuel in the world, mainly due to Brazil's decision to produce fuel alcohol from sugarcane. As a result of this, for many years Brazil was the world's main user of ethanol as a fuel. According to industry sources, the USA has surpassed Brazil in the production and use of ethanol. In 2003, US ethanol production capacity was 12.2 billion litres, of which 7.9 billion litres was consumed as fuel ethanol (Yacobucci and Womach, 2003) whereas, total ethanol production in Brazil according to <http://www.unica.com.br/i_pages/estatisticas.asp#>, was 6.95 billion litres.

In many countries the development and use of alcohol fuels in transport have, for the most part, been driven by the desire to find renewable substitutes for imported petroleum-based fuels. Alcohol fuels have also been used as additives to conventional fuels to improve fuel characteristics. For petrol blends, ethanol is a known octane enhancer (a component added to petrol to increase octane rating and reduce engine "knock" - refer Section 8.1.2) and oxygenate (a fuel or fuel additive containing hydrogen, carbon and oxygen in its molecular structure – refer Section 8.1.2). Ethanol easily blends with gasoline but blending with diesel requires an emulsifier or additive to form a stable fuel.

More recently alcohol fuels have been the focus of attention as a possible means of reducing greenhouse gas emissions, and noxious urban emissions from transport. Results from several studies that have been conducted throughout the world on exhaust emissions from ethanol blended fuels are often contradictory, making it difficult to generalise on emission outcomes and performance of ethanol blends. The difficulty is compounded by the fact that tailpipe emissions, and to a certain extent evaporative emissions, vary markedly depending on the exact nature of the petrol with which the ethanol is blended and the exact nature of the vehicles that use the fuel. In addition, the air pollution potential of the resulting emissions depends on the exact nature of the airshed.

There has been substantial US interest in the use of ethanol in cars. The reason for this stems from the US Federal Reformulated Gasoline (RFG) Program that was mandated by the Clean Air Act Amendment of 1990. The RFG Program attempts to lower motor-vehicle emissions through re-engineering petrol blends. For example, the Clean Air Act mandates a specified minimum oxygen content in RFG blends. By itself, conventional petrol has no oxygen content. Therefore oxygen-containing chemical additives, known as oxygenates, are blended into the fuel (refer Section 8.1.2). Originally such reformulated gasoline was made by blending MTBE (methyl tertiary butyl ether) into petrol. Because of the contamination of Californian groundwater with MTBE the Californian Governor ordered the removal of MTBE from petrol and studies on the environmental and health effects of ethanol in petrol. The use of ethanol produces an oxygenated fuel that satisfies the requirements of Californian reformulated gasoline. Reuter *et al.* (1992) studied European petrol oxygenated with MTBE, ETBE and ethanol and found that the emissions of oxygenated petrol are independent of the oxygenate that is used. News reports indicate that Japan is also considering adding ethanol to fuel.

Ethanol can be produced in two forms — hydrated and anhydrous. Hydrated ethanol has a purity of 95% suitable for blending with an ignition improver, or as a 15% emulsion in diesel that is known as Diesohol. Hydrated ethanol, sourced from the CSR Yarraville distillery in

Melbourne, is used as the fuel in the three “sugar buses” that Ventura Bus lines operates around the eastern suburbs of Melbourne.

A second stage refining process is required to produce anhydrous ethanol (100% purity) for use in ethanol blends in petrol. Most industrial ethanol is denatured (to prevent oral consumption) by the addition of small amounts of an unpleasant or poisonous substance.

Anhydrous ethanol can be used as an additive in petrol, or as a fuel in its own right. Despite this, as an automotive fuel in Brazil it is usually composed of 85% ethanol with 15% petrol (E85). The addition of 15% petrol improves the ignitability of the alcohol, especially at low temperatures. Other additives have also been trialled as ignition improvers. Petrohol (petrol and ethanol blends that range from 5% to 26% ethanol) consists of a blend of anhydrous ethanol and petrol. This study will use the term petrohol (or E10) to refer to any grade of petrol blended with a 10% ethanol blend. Such fuel has an oxygen level of 3.5%.

Both BP and Caltex have, at various times, conducted trials of E10 fuels in Queensland. In such cases the material has been shipped from Queensland to Victoria for processing at the Yarraville refinery, then re-shipped back to Queensland.

8.1.2 Characteristics of Alcohol Fuels

Ethanol (C_2H_5OH) is an alcohol, an oxygenated organic carbon compound and contains 34.7% by weight of oxygen. It is the intoxicating component of alcoholic beverages and is widely used in the chemical and pharmaceutical industries. By contrast, petrol and diesel are both mixtures of a range of hydrocarbon compounds, none of which contain oxygen and these compounds have significantly different chemical and physical properties than those of alcohols (API, 2001). The addition of ethanol to petrol or diesel (in blended fuels) changes a number of important fuel characteristics including combustion properties (octane enhancement), energy content, and vaporisation potential.

The octane enhancing effect of ethanol is due to its high latent heat of vaporisation (cooling of the fuel charge in the combustion chamber) and high combustion expansion ratios (API, 2001). Ethanol, therefore, can lead to greater fuel efficiency, but this is offset by the characteristic lower fuel energy content.

8.1.2.1 Fuel Energy Content

The energy contents of ethanol fuel blends are lower than for the base fuels. The energy content[6] of pure ethanol ranges from 21 MJ/L (LHV) to 23 MJ/L (HHV). This compares to 30.5 MJ/L (LHV) and 35 MJ/L as representative values for petrol, and 38.6 MJ/L for diesel. The energy content of ethanol depends on whether it is in the hydrated or anhydrous form.

8.1.2.2 Fuel Octane Rating

The octane rating of a fuel refers to the tendency of the fuel to self-ignite (knock) during engine combustion. Engine knock occurs when the fuel ignites in the compression stroke prior to application of the spark and can lead to severe engine damage. In order to compare the octane quality of different fuels, the fuels are combusted in a test engine relative to the fuel iso-octane, which is arbitrarily given an octane rating of 100. The higher the octane rating, the greater is the knock resistance of the fuel and the higher is the compression ratio that can be used resulting in

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[6] Low Heating Value (LHV) is more relevant than the High Heating Value (HHV) for comparisons between vehicle fuels.

increased engine efficiency. Normally, the octane rating is given as the Research Octane Number (RON) and Motor Octane Number (MON). The RON is related to city driving and is measured at low speed in the test engine (600 rpm), whereas MON is related to highway driving and is determined at higher speeds (900 rpm). RON is usually greater than MON and the difference between them, called fuel "sensitivity", can be used as a measure of fuel quality.

Ethanol is one of a number of compounds that may be used in gasoline as an oxygenate and octane enhancer. Other compounds that may be used to enhance octane include MTBE, ETBE and methanol. Among the alcohol compounds that may be used to enhance octane, DSA (2000) notes that ethanol and iso-propyl alcohol (IPA) are among the best for raising existing 91 RON rated gasoline to 95 RON rating. DSA (2000, Table 4, p. 19) provides some estimates of the impact on RON of adding varying concentrations of octane enhancing compounds to 1998 pool gasoline, with an initial octane rating of 91.6/82.5 (RON/MON). Ethanol and IPA provide the biggest lift in RON per unit volume of additive, lifting the RON to 93.9 and 94.1, respectively.

Some additional refining, to lift the RON of the base petrol stock would be required to ensure that the ethanol blend fuel obtained 95 RON[7]. Ethanol lifts the oxygen content to 3.5% by weight, which is at the limit allowed for ethanol blended fuels under the current fuel standards. Until now, Australian refineries have generally used refinery processes to convert low octane components into higher ones, thereby enhancing the overall octane rating of the fuel. Ethanol has probably not been used by Australian refiners to enhance octane because of price and reliability of supply.

In many countries, MTBE is the preferred oxygenate of the oil industry. MTBE is used widely in Europe and Asia (DEH, 2000) to enhance octane rating. MTBE is produced by the addition of methanol to olefin isobutene, a by-product of refinery and petrochemical operations. Typically MTBE can be blended with petrol in concentrations of up to 15%[8].

Concerns about the environmental effects from the leaching of MTBE into groundwater have led to a ban on the use of MTBE in transport fuels in some States of the US and to Australian standards that will effectively ban fuel containing MTBE in Australia from 1 January 2004. The problem concerning the use of MTBE in petrol is fundamentally related to the failure to properly transport and store petrol. DSA (2000) notes that it is unlikely that Europe could adopt the new Euro III and Euro IV emissions standards without the use of MTBE in gasoline.

Ethanol is probably one of the few alcoholic or ether derived oxygenates currently permitted for blending with gasoline. Under the current fuel standards, DIPE and TBA are effectively prohibited from use in petrol in Australia, as will MTBE be from 1 January 2004. And, according to DSA (2000), as ETBE, TAME and ETAE were not then listed on the *Australian Inventory of Chemical Substances*, consequently they were not permitted to be used in Australia.

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[7] DSA (2000, Table 5, p. 20) reports that the base fuel stock, required for 10% blending with ethanol to produce an E10 blend at 95 RON with an RVP of 60 kPa, would have RON of 92.8 and RVP of 41.5. Some additional refining processes would be required to produce such a base petrol stock.

[8] Prior to the ban on the use of MTBE in the US, the California Energy Commission report (CEC, 1999) noted that MTBE concentrations were typically around 11%, by volume, of US gasoline. The California Energy Commission (1999) study also concluded that ethanol was the most expensive of the alternative oxygenates, increasing the average cost of gasoline by up to 6.7 (US) cents per gallon (or approximately 5%).

8.1.2.3 Fuel Volatility

Fuel volatility characteristics (vapour pressure, distillation profile[9] and vapour to liquid ratio[10]) are important for vehicle emissions and driveability. Fuel volatility is controlled during fuel production to meet the specific climate requirements of the local market.

The ADR37/00 specifies fuel volatility in terms of Reid Vapour Pressure[11] (RVP) with a higher value of RVP corresponding to greater fuel volatility. Ethanol has a low RVP (16.5 kPa) but, due to the polar nature of the ethanol molecule, vapour pressure anomalies occur when it is added to non-polar hydrocarbon-based conventional fuels. Ethanol increases the vapour pressure of the fuel blend but not in proportion to the ethanol vapour pressure or concentration. Fuel RVP increases as ethanol is initially added with the greatest RVP at an ethanol content of approximately 5% (v/v) (RVP increase ~ 1 psi or 6.9 kPa) and for concentrations greater than 5% (v/v), the RVP slowly decreases (NRC, 1999).

The impact on higher fuel RVP and increased evaporative emissions from the addition of 10% (v/v) of ethanol is described in Section 8.3.3.2.

8.2 Upstream Emissions from Use of E10

8.2.1 Production and Distribution

8.2.1.1 Ethanol Production

Ethanol can be manufactured from biomass via:
- fermentation of sugar derived from grain starches or sugar crops; and
- utilisation of the non-sugar lignocellulosic fractions of crops.

The details of the manufacturing process of ethanol are given in Beer *et al.* (2001). The calculations used in this report are taken largely from the same data developed in the CSIRO *Comparison of Transport Fuels* study (Beer *et al.*, 2001) with a number of modifications:
- a credit for the production of bio-dunder;
- transport emissions for distribution of ethanol; and
- change in the production replacement for molasses.

8.2.2 Bio-dunder Allocation

Dunder or bio-dunder from molasses based ethanol distilleries has traditionally been a source of pollution, but has more recently been used as a fertiliser, or as an additive to a fertiliser mix. Bio-dunder is used for its potassium content (Nutri-Tech, 2002), which is around 3-4% and is also used for its nitrogen content, which is assumed to be around 0.5%. This reduces the need for potassium fertilisers such as muriate of potash (which has a potassium content of 50%) by 80g/kg of bio-dunder, and urea (which has a nitrogen content of 45%) by 11g/kg of bio-dunder.

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[9] The distillation profile of ethanol blended fuels is affected by the addition of 10% (v/v) ethanol to petrol, with respect to depressing the mid section of the fuel distillation curve, which results in decreased driveability (API, 2001).

[10] Vapour to liquid ratio refers to the potential for fuel vapour lock from highly volatile fuels at elevated operating temperatures, and which increases evaporative emissions.

[11] In the US, the RVP method of measurement of vapour pressure may be superseded by other techniques that eliminate the possibility of water contamination during testing (ethanol has a high affinity for water) (API, 2001).

8.2.3 Transport Emissions for Distribution of Ethanol from Molasses

It is difficult to determine what, if any, additional transport would be required for the use of 350 ML of ethanol in Australia in E10.

The Australian Institute of Petroleum (AIP) (1997) suggests approximately 19% of petroleum products in Australia are shipped around the Australian Coast (refer Table 9). This suggests that around 80% of petroleum products are manufactured in capital cities of the States where they are sold.

Table 9. Transport of petroleum products around Australia in coastal shipping

1994/95	**Petroleum (kt)**
Vic	1,965
NSW	577
WA	1,342
SA	691
Queensland	1,747
Total transported	6,322
Total production 1994/5 (ML)	42,665
Tonnes (estimate)	34,132
Percentage Transported	19%

Table 10 details some assumptions about the production and distribution of ethanol for this analysis. It starts with the assumption that the present system whereby molasses is shipped from Queensland to Victoria, and ethanol is shipped from Victoria to Queensland will be discontinued. It is assumed that fuel ethanol from sugar is produced from around Sarina in Central Queensland and then assumes that this will be distributed equally through each of the three eastern seaboard States, based on their current consumption of petrol. It also assumes intra-State transport by road and interstate transport by rail. The values in Table 10 indicate average road transport haulage of 700 tonne km per tonne, and average rail haulage of 1903 tonne km per tonne.

Table 10. Sales and transport assumptions for ethanol produced from molasses in central Queensland

	Market Share by State (%)*	ML consumption by State based on 15000 ML	E10 sales by State for 3500 ML‡	Distance from Central Queensland Coast km	Road Transport ktonne km	Rail Transport ktonne km	Comment
Vic	25	3,750	1,182	3,000		2,802,365	Assume rail
NSW	29	4,350	1,372	1,900		2,058,804	Assume rail
QLD	20	3,000	946	700	662,162		Assume road
SA	8	1,200	0	2,800			
WA	13	1,950	0				
Tas	2	300	0	2,400			
NT	3	450	0	1,800			
Total		15,000	3,500		662,162	4,861,169	

*Based on Australian Institute of Petroleum (1997).
‡Note E10 sales are assumed to be equally spread through Queensland, NSW and Victoria at 32% of petrol sales.

8.2.4 Change in Production Replacement for Molasses

In the original CSIRO study (Beer *et al.*, 2001) the expanded system boundary allocation for molasses assumed the molasses used in fuel production would need to be replaced by low-grade wheat products and wheat starch in fermentation processes including monosodium glutamate production. These low-grade wheat-based materials are also used in stockfeed for their protein and energy content and they were assumed to be replaced by soybeans as a protein source. On reviewing this, it is not likely that soybeans would be used in this instance, although soy meal certainly is, but this again is another constrained co-product limited in part by demand for soy oil. More basic crops that are grown specifically for fodder (and for soil nitrogen replacement) are lupins and field peas. Lupins have a protein content of around 35% and energy content around 19%. Wheat has an energy content around 18% but a protein content around 12%. With equivalent energy content, but three times the protein, the ratio of lupins required to replace wheat-based material is taken to be approximately 1:2 lupins to wheat material.

8.2.5 Wheat (Primary) Ethanol via Fermentation in a 40 ML Plant

This scenario is taken largely from the data provided in the CSIRO *Comparison of Transport Fuels* study with a number of modifications, such as:

- an estimate of the environmental impacts from capital construction (see the previous section on ethanol from molasses for capital cost data);
- an examination of benefits of wheat straw;
- transport emissions for distribution of ethanol.

8.2.5.1 Wheat Straw Allocation

Wheat straw can be used as mulch and nutrient input cropping, stockfeed, or a building material. As mulch it provides many benefits in water retention, reducing erosion and increasing soil depth. It can also add nitrogen, potassium and phosphate. As it is likely to be used within wheat production, and it is unclear whether this use is already accounted for in the fertiliser use data gathered on primary wheat production, no co-production benefits will be provided for wheat straw.

8.2.6 Transport Emissions for Distribution of Ethanol from Wheat

In the same way that transport of ethanol from molasses was allocated in the previous section, Table 11 details the assumed production and distribution of ethanol from wheat, and the assumed transport distances. It is assumed that all 350 ML of fuel ethanol is split between the major Wheat-growing States. Distribution distance is then taken from the State's wheat belt area to the capital city. Due to the small transport distances, road transport is assumed. The data indicate average road haulage of 272 tonne km per tonne.

Table 11. Transport emissions for distribution of ethanol from wheat sources

Market	Market Share by State (%)	Consumption by State (ML)	E10 sales by State	Distance from ethanol production‡ (km)	Road Transport (tonne km)
Vic	25	3,750	1,167	400	368,667
NSW	29	4,350	1,353	400	427,653
Queensland	20	3,000	0	0	
SA	8	1,200	373	200	58,987
WA	13	1,950	607	200	95,853
Tas	2	300	0	0	
NT	3	450	0	1,800	
Total		15,000	3,500		951,160

*Based on Australian Institute of Petroleum (1997).
‡Production assumed to be split between the major wheat growing States (Vic, SA, NSW and WA)

8.2.7 Wheat Starch Wastes Ethanol via Fermentation in a 40 ML Plant

Starch waste streams have traditionally been a problematic waste discharge to waterways because of their significant environmental impact. Such discharge is no longer legal, and is unlikely to be economically sensible as suggested by data on the economics of starch production shown in Table 12. It shows the value of waste products from starch production to be between 3–4% of the total production costs. While these data relate to potatoes and cassava-based starch, similar results could be expected for wheat-based starch production.

In this assessment, using the system boundary expansion approach, alternative uses for the starch are taken as stockfeed formulation. However, it is difficult to estimate the value of wheat starch waste to stockfeed. To produce ethanol the starch waste must have fermentable starches and sugars. It takes around 2.1 kg of wheat containing 52% starch to produce a litre of ethanol, and then the starch content in waste starch products must be close to 1 kg per litre of ethanol produced. However, the concentration of starch and combinations of other content in the waste stream may make it unsuitable for higher-grade uses such as stockfeed. With this level of uncertainty and given the recent history of the material as a pollutant, no environmental load has been allocated to the wheat starch waste[12].

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[12] It is standard practice in life cycle analysis to assign zero environmental load to waste products. This results in favourable environmental benefits being assigned to products made using waste material.

Table 12. Economics of starch production in three Asian plants

	Thailand cassava starch mill		Indonesia cassava starch mill		China sweet potato starch mill	
	5000 tons/year		8000 tons/year		5000 tons/year	
Capacity	**Unit price (US$/ton)**	**Proportion (%)**	**Unit price (US$/ton)**	**Proportion (%)**	**Unit price (US$/ton)**	**Proportion (%)**
Raw material price	35		40		35	
Raw material cost	175		200		233	
Value of waste products	-9.2		-8.7		-9	
Raw material net cost	165.8	76	191.3	70	224	80
Capital and labour cost	52	24	82	30	56	20
Native starch cost	217.8	100	273.3	100	280	100
Waste value as % of starch costs		4.2		3.1		3.2

Source: (Fuglie and Oates, 2002). The data are based on the US short ton.

8.2.8 Differences Between Wheat Ethanol and Sweet Sorghum

As there is no plant operating in Australia using pure wheat or sweet sorghum, it is difficult to estimate exactly how ethanol production from these crops would differ. The data in this study is based on a range of literature and overseas studies, particularly in relation to sweet sorghum.

The Queensland government broad acre field crop website suggests that sorghum usually yields better than maize, and the fertiliser requirements are similar, with 60-180 kg of N per ha compared with around an average of 4 kg/ha for wheat. Yields of useable by-products (distiller's grains) from sorghum processing into ethanol are much larger than for wheat, and the benefits derived from these, give sorghum marginally lower greenhouse impacts per litre of ethanol produced.

8.3 Downstream Emissions from Use of E10

8.3.1 E10 Fuel Properties

The addition of 10% (v/v) ethanol to conventional petrol affects a number of properties of the fuel blend (refer Section 8.1.2). Table 13 shows the fuel properties of the base fuels (LP and ULP) and the E10 blended fuels used in the Australian Petrohol study as well as the mean fuel consumptions calculated from the two vehicle groups using both E10 and the relevant base fuel (APACE, 1998). Ethanol increased the octane rating of the fuels — RON was more significantly increased than MON. Although ethanol increased the octane benefit of the fuels, the difference between RON and MON was also increased from about 9 to 11 octane units.

The volatility of both fuels was significantly increased (about 6%) by the addition of ethanol, which had the negative effect of increased evaporative HC emissions (refer Section 8.3.3.2).

The addition of ethanol to petrol increases combustion efficiency due to the oxygenate effect but this is offset by the decrease in energy content of the E10 blended fuels relative to the base fuels and results in increased fuel consumption of approximately 2%. The fuel consumptions for the "City cycle" and "Highway cycle" were calculated using the measured CO_2 emission, from the ADR37/00 emission tests of all vehicles, according to Australian Standard AS2877 —

"Methods of test for fuel consumption of motor vehicles designed to comply with Australian Design Rules 37 and 40". Motor vehicle manufacturers report a vehicle's fuel consumption using the National Average Fuel Consumption (NAFC) ratio (weighting of 55 City / 45 Hwy).

Table 13. Fuel consumption and fuel properties of petrol and petrohol (E10) fuelled Pre- and Post-1986 vehicles (Petrohol study)

	Mean Fuel Consumption (L/100km)[a]			**Fuel Density (kg/L)**[b]		**RON**	**MON**	**RVP (kPa)**[b]
	City Fuel	**Hwy Fuel**	**NAFC**[c]	**10°C**	**15°C**			
PRE-1986 Petrol	12.6	9.1	11.0	0.7369	0.7406	96.0	86.7	82.3
PRE-1986 Petrohol	12.7	9.3	11.2	0.7324	0.7361	98.7	87.6	87.4
POST-1986 Petrol	12.1	8.8	10.6	0.7329	0.7376	91.9	82.9	80.6
POST-1986 Petrohol	12.5	9.0	10.9	0.7284	0.7331	95.5	84.1	85.4

[a] Fuel consumption calculated from CO_2 measurement according to AS2877
[b] Batch fuel
[c] National Average Fuel Consumption (NAFC) calculated using 55/45 City/Hwy ratio

8.3.2 E10 Tailpipe Emissions

Motor vehicles emit a range of tailpipe or exhaust emissions (regulated pollutants, GHG, air toxics, particulate matter as well as secondary pollutants). Exhaust emissions are dependent on a wide range of variables including: driving patterns, fuel type, as well as various vehicle and engine specific factors such as design, size, state of tune and type and condition of emission control systems.

The ability of ethanol to contribute to a reduction in greenhouse gas emissions on a FFC basis is very much influenced by the nature of the feedstock and by the source of power used for the production process. CO_2 emissions from the combustion process alone are fairly similar for alcohol fuels and gasoline on an energy equivalent basis, assuming complete combustion.[13] The tailpipe emissions for the ethanol (10% v/v) based fuels petrohol (E10 ULP and E10 LP) are discussed below and LCA analyses of each of the fuels are detailed in Chapter 10.

8.3.2.1 Vehicle Emission Studies

There have been very few E10 vehicle emission studies conducted that are relevant to Australian conditions (refer to Appendix I). The only significant Australian study into the emissions effect of adding E10 in the Australian vehicle fleet was conducted by APACE Research (APACE, 1998).

APACE Petrohol Study
As part of this study, the NSW EPA conducted the "Petrohol In-Service Vehicle Emission Study", in which they tested 60 in-service light-duty passenger vehicles over a 2-year period from 1995 to 1997. All vehicles were tested according to the relevant Australian Design Rule (ADR37/00), and a total of 188 complete emission tests were conducted in this program. The vehicles were selected from five different manufacturers with the major vehicle makes and

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[13] Emissions of CO_2 from ethanol are 64.4 grams per MJ, and from diesel 69.7 grams per MJ based on the HHV. Emissions of CO_2 from the combustion of one litre of fuel are 1.5 kilograms for ethanol, and 2.7 kilograms for diesel.

models (1979 to 1995) being representative of the in-service fleet. The vehicles were all selected from the Newcastle/Central Coast region of NSW and it was not established if this vehicle sample is representative of the Australia wide in-service fleet. The vehicle sample covered both carburettor and fuel injection type vehicles with a range of odometer readings (7,000 to 440,000 km). The 60 vehicles tested were grouped into 19 leaded fuelled vehicles (pre-1986 models) and 41 unleaded fuelled vehicles fitted with a catalytic converter (post-1986 models). The post-1986 (includes 1986 year model) model vehicles contain either a 2-way or 3-way converters. The proportion of 2-way and 3-way catalysts (TWC) is not known, however, TWC do predominate (NGGIC, 1998).

Of the 60 vehicles tested, 37 were tested once (Base Fleet vehicle category) following standard engine tuning. Two aims of the study were to determine the effect of maintenance in reducing emissions and determine the deterioration in emissions over a 12-month period. A category of vehicles, Long Time In-Service (LTIS) (11 vehicles), was tested 3 times: post tune (like Base Fleet), and 12 months later pre-tune and post-tune testing.

It should be noted that the ADR37/00 is equivalent to the US Federal Test Procedure (FTP) and may not represent accurately the real on-road driving patterns. The actual on-road emissions are expected to be higher than predicted by the ADR cycle test due to the fact that the ADR (and FTP) drive cycles tend to have (Watson, 1995):

- lower acceleration rates and speeds;
- longer idle times; and
- an early and relatively high speed event leading to the catalyst reaching operating temperature relatively quickly.

AQIRP

The most thorough overseas fuel oxygenates study was conducted by the US Auto/oil Air Quality Improvement Research Program (AQIRP), which commenced a comprehensive analysis of fuel oxygenates in a variety of fuel types in 1989. Two fleets of well maintained vehicles: "current" (10 pairs of 1989 model vehicles) and "older" (7 pairs of 1983-1985 model vehicles) technology were used. Federal test procedures were employed to measure exhaust and evaporative emissions (mass and speciated). E10, one of the fuels tested, was blended using 4 different base gasolines. One of the tests investigated the change in evaporative emissions when the RVP of an E10 blend was reduced by 6.9 kPa (refer Section 8.3.3). Results from the AQIRP program have been summarised (API, 2001) and reported in a number of publications (AQIRP, 1993a, 1993b, and Colucci and Wise, 1992)

8.3.2.2 Greenhouse Gases (GHG)

Carbon Dioxide (CO_2)

Of the greenhouse gases emitted in vehicular exhaust, carbon dioxide is by far the largest mass emission. The tailpipe emissions of CO_2, the main GHG, show increases of between 1.8 and 4.8 g/km of CO_2 for vehicles using petrohol, relative to the base petrol (refer Table 17).

Tailpipe emissions studies of the next most significant greenhouse gases CH_4 and N_2O are less common (especially true for the Australian transport sector) and can show large variations in results.

N_2O and CH_4 are more potent GHG than CO_2 on a per molecule basis. An international agreement is used to weight IPCC 100 year global warming potentials (GWP) as given in Table 14.

Quantifying N_2O and CH_4 emissions is difficult due to the many variables involved. The tailpipe emissions of these gases, N_2O in particular, are influenced to a large degree by the pollution control systems (type, age, condition) employed by the vehicles being tested. The only reliable way to estimate the emissions of these two GHGs is by using emissions measurements from the vehicle fleet with each combination of fuel, vehicle engine technology, combustion environment and emission control systems being tested.

Table 14. IPCC 100 Year global warming potentials (GWP)* of transport GHG

Gas	GWP
Carbon dioxide CO_2	1
Methane CH_4	21
Nitrous oxide N_2O	310

* See http://unfccc.int/resource/docs/cops/07.pdf for explanation of this weighting

Nitrous Oxide (N_2O)

Significant uncertainty exists when it comes to quantifying fleet N_2O emissions because N_2O emissions are dependent on catalyst factors (composition, age, and degree of deactivation) and driving cycle variables. Based on very limited data (and the primary role of TWC in N_2O production), the N_2O emission factors would not be expected to be significantly affected by E10. A minor impact of E10 could potentially result from the decreased sulfur content of fuel (from volumetric dilution of sulfur-containing petrol) as high fuel sulfur has been found to be influential in the increase of N_2O, due to the decreased effectiveness of NO_x conversion by the catalyst (Schifter *et al.*, 2001 and references therein).

Effect of Catalytic Converter

N_2O vehicle emissions are a complex function of the type and condition of the emission control system more so than being related to total NO_x levels. The type of drive cycle impacts on N_2O emission levels as well, with higher speeds (highway) and incline gradients leading to lower emissions (can be explained largely by effect on catalyst temperature). When comparing the real world emissions with the FTP, N_2O emissions are higher on account of the additional cold starts (underestimated in the FTP), but lower for the higher speeds.

The bulk of the vehicle N_2O emissions are produced by the exhaust catalyst, and not from engine combustion — the high temperatures and pressures (also dependent on the air/fuel ratio) in IC engines produce NO_x but seem to be inefficient in producing N_2O. Vehicles not fitted with a catalytic converter, therefore, essentially produce no N_2O. The effect of catalytic converters on N_2O vehicle emissions is discussed in more detail in Appendix I.

N_2O Emission Factors

A number of internationally recognised organisations including the US EPA and IPCC have attempted to estimate the emission factors of N_2O and CH_4, however, the data relied upon to do so is generally insufficient. The revised IPCC (1997) guidelines on GHG emissions state that, for alternative fuels, N_2O emission factors are “not available”. The US EPA (2003b) GHG Inventory determined the N_2O emission factors for conventional fuel type vehicles based on a series of vehicle emission tests (for EPA Tier 1 and LEV vehicles), conducted at its National Vehicle and Fuel Emissions Laboratory (NVFEL), combined with a literature review (Michaels, 1998).

In Australia the BTRE (2002) view the N_2O emission factors (for catalyst equipped vehicles) from the National Greenhouse Gas Inventory (NGGI) process (NGGIC, 1998) as too high (based on IPCC 1995 Volume 3) and use instead emission rates previously presented by the Bureau (BTCE 1995) and the US EPA.

N_2O Emission Factors for Biofuels

To estimate N_2O emission factors for alternative fuel types, the approach taken by the US EPA (2003b) GHG inventory is to use the NO_x/N_2O ratio, using NO_x from vehicle emission tests. A NO_x/N_2O ratio of 5.7 is used for EPA Tier 1 cars and Lipman and Delucchi (2002) found NO_x/N_2O of 3 to 5.5 for older technology vehicles.

Lipman and Delucchi (2002) analysed a database of emission measurements of a variety of different vehicle technologies and fuel types (fuel types did not include E10). They estimated emission factors for CH_4 and N_2O and compared them to recent US EPA (US EPA, 1999) and IPCC (IPCC, 1997) estimates. The major N_2O findings of this study, with respect to ethanol (not petrohol) fuelled vehicles, are:

- based on the limited data, ethanol fuelled vehicles emit similar levels of N_2O as conventional petrol fuelled vehicles;
- developed N_2O emission factor calculations for Tier 0 (early catalyst technology) and Tier 1 (advanced catalyst technology) vehicles utilising catalyst deterioration factors are based on linear regression analysed data and using average vehicle life of 70,000 miles (equivalent to 112,654 km which may be considered lower than the expected average vehicle life of approximately 150,000 km in Australia);
- a range of N_2O emission factors for non-catalyst equipped vehicles — from 0-0.01g/km; and
- diesel vehicles appeared to emit N_2O to a similar degree to petrol vehicles — although likely to be at a marginally lower level (limited by data).

The N_2O emission factors are represented by the following:

$$EMT0/T1 = ZMT0/T1 + DA * (Miles/1000)$$

EMT0/T1 is the N_2O emissions from Tier 0 or Tier 1 petrol light-duty passenger vehicles (g/mile), DA is the deterioration rate in emissions with vehicle mile and Miles is the total mileage of the vehicle. DA is 0.00136 and 0.00016 g/mile per 1000 miles, respectively for Tier 0 and Tier 1 vehicles. ZMT0/T1 is the N_2O emissions from new Tier 0 or Tier 1 petrol light-duty passenger vehicles (g/mile).

The US classed Tier 0 (employing early catalyst technology) and Tier 1 (advanced catalyst technology) could be expected to represent present and future (e.g. 2010) fleet average catalyst equipped vehicles respectively. The calculated Tier 0 emission factor was 0.078 g/km, with Tier 1 considerably lower at 0.033 g/km. These estimated values can be compared to the lower values given in the US inventory of GHG emissions and sinks for the period 1990-1999 (refer Table 15). The higher emission factors from Lipman and Delucchi were used in this study, in preference to the US inventory values due to the greater age (and expected higher emissions) of the vehicle fleet in Australia, relative to the US. The magnitude of the N_2O emissions factor is of secondary importance; however, as the E10 emission factor for N_2O for petrohol was assumed to be unchanged from conventionally fuelled vehicles. This was further supported by a vehicle emission study (5 vehicles ranging over all emission control technology types and tested over FTP cycle), which showed no statistically significant difference in N_2O emissions between E10 and conventional petrol (Warner-Selph and Harvey, 1990). Under real world conditions, many potential confounding effects could influence the impact of E10 on N_2O emissions. It is possible that the addition of ethanol to petrol could potentially result in increased N_2O emissions as a result of any increased NO_x emissions produced under instances of open-loop engine operation (due to the "leaning effect" from the oxygenate). Addition of 10% (v/v) of ethanol would also cause a volumetric dilution of known petrol derived catalyst contaminants (such as sulfur, phosphorous and lead).

Table 15. CH_4 and N_2O emission factors (g/km) from US inventory of GHG emission and sinks: 1990-1999

US Emission Standard	N_2O	CH_4
Non-catalyst	0.010	0.12
Tier 0	0.051	0.04
Tier 1	0.029	0.03

Source US EPA (1999)

Methane (CH_4)

CH_4 emissions occur due to incomplete fuel combustion in the engine combustion chamber and are relatively higher with fuels of lower aromatic contents.

CH_4 emissions are also dependent on factors related to the catalytic converter — CH_4 can be produced in catalytic converters from incomplete catalytic HC oxidation (Poulopolous *et al.*, 2001). CH_4 is difficult to catalytically oxidise (especially at low temperatures) and therefore CH_4 emissions would presumably increase with catalyst age/deterioration. It could, therefore, be anticipated that CH_4 emissions could be higher when the catalyst is not at full operating temperature. There would be an expected correlation between CH_4 and NMHC emissions but a higher fraction of CH_4 emissions (of THC) would be likely, however, from more modern emission controlled technologies.

As part of the NSW EPA Petrohol study, CSIRO conducted limited (2 pre-1986 and 9 post-1986 vehicles) HC speciation for CH_4 solely on the LTIS group of vehicles. A good correlation was found between the NSW EPA and CSIRO THC measurements (approximately 1:1) (APACE, 2003a). Using this correlation, and the CSIRO CH_4 ratio to THC, APACE estimated CH_4 emissions for ULP and petrohol (post-1986 vehicles only) with respect to: city and highway cycles, and National Average Fuel Consumption (NAFC) weighting (55 City /45 Hwy). The CH_4 emission estimates (city cycle) for ULP and petrohol (post-1986 vehicles) were 0.0650 and 0.0577 g/km, respectively.

APACE estimated CH_4 for various vehicle groupings from the correlation they found between CH_4 and HC. This correlation combined ULP and LP data together. Analysis of the CSIRO HC speciation data, however, shows a difference between the CH_4/HC emissions ratios of LP and ULP vehicles. These differences indicate a catalytic converter effect, of increasing the CH_4 ratio to the HC emissions (sum of CSIRO speciated HC), by selectively catalytically oxidising the NMHC, relative to CH_4. The pre- and post-1986 vehicles showed average CH_4/HC mass emission ratios of 0.154 and 0.067, respectively. Analysis of the data, regarding the specific stage of the ADR37/00 cycle being sampled, showed that all stages of the ADR cycle with the highest CH_4/HC ratios were all from post-1986 vehicles and the highest ratio was from Bag 2 (stabilised stage where catalyst would be expected to be at full operational temperature), as shown in Table 16. The CH_4/HC ratio for post-1986 vehicles at each stage of the ADR cycle showed greater variability (range 0.088 to 0.224) than that of the pre-1986 vehicles. The CH_4/HC ratio for pre-1986 vehicles at each stage of the ADR cycle was consistent (range 0.044 to 0.072). In addition, the effect of ethanol addition to the petrol was decreased CH_4 emissions (and decreased THC). This effect was significantly greater for non-catalyst vehicles (39.1% CH_4 reduction) than for the catalyst-equipped vehicles (8.5% CH_4 reduction). This is further supported by the significantly greater CH_4 emission reduction (16.4%) in the cold start (Bag1) ADR phase for the catalyst-equipped vehicles.

Table 16. Methane (CH_4) vehicle emissions and CH_4/HC ratios from Pre- and Post-1986 vehicles with petrol and petrohol (in decreasing order of CH_4/HC ratio)

Vehicle group	ADR PHASE[a]	Fuel	NMVOC (mg/km)	HC[b] mg/km	CH_4 (mg)/km	↓CH_4 from E10 (%decr)	CH_4/ HC ratio
POST-1986	stabilised	E10	172	222	49.8	5.1	0.224
POST-1986	stabilised	ULP	261	314	52.4		0.167
POST-1986	Hot	E10	214	255	40.8	7.5	0.160
POST-1986	AV	E10	266	315	48.5	8.5	0.154
POST-1986	Hot	ULP	248	292	44.1		0.151
POST-1986	AV	ULP	339	392	52.9		0.135
POST-1986	cold start	ULP	649	714	65.4		0.092
POST-1986	cold start	E10	567	622	54.7	16.4	0.088
PRE-1986	stabilised	LP	1699	1830	131.0		0.072
PRE-1986	cold start	LP	1933	2078	144.6		0.070
PRE-1986	AV	LP	1694	1817	123.3		0.068
PRE-1986	Hot	LP	1505	1607	101.8		0.063
PRE-1986	cold start	E10	1689	1796	106.9	26.1	0.060
PRE-1986	stabilised	E10	1273	1345	71.7	45.8	0.053
PRE-1986	AV	E10	1352	1427	75.1	39.1	0.053
PRE-1986	hot	E10	1247	1305	57.7	43.3	0.044

[a] "cold start" =bag1; "stabilised"=bag2; "hot"=bag3; "AV"=ADR average 3 bags
[b] HC =NMVOC+CH4

Lipman and Delucchi (2002) estimated emission factors for CH_4 and compared them to recent US EPA (US EPA, 1999) and IPCC (IPCC, 1997) estimates. The major findings of this study, with respect to ethanol (not petrohol) fuelled vehicles, are discussed in Appendix I.

Poulopolous *et al.* (2001) found in their engine dynamometer study that CH_4 engine emissions were the same or lower for E10 compared with the conventional fuel, however; the tailpipe emissions (Pt/Rh TWC) showed lower CH_4 using E10 at lower relative engine power, and higher CH_4 using E10 at higher engine power conditions, compared with the base petrol.

Table 17. Estimated tailpipe emissions factors (g/km) of greenhouse gases and gross CO_2-e from Pre- and Post-1986 classed vehicles with petrol and petrohol

	CH_4[c]	N_2O	CO_2	Gross CO_2-e
PRE-1986 Petrol	0.123	0.012[a]	255.1	261.4
PRE-1986 Petrohol	0.075	0.012[a]	259.9	265.2
POST-1986 Petrol	0.053	0.078[b]	264.4	289.6
POST-1986 Petrohol	0.048	0.078[b]	266.2	291.3

[a] US vehicles no pollution control system not tested with petrohol
[b] US vehicles Tier 0 catalyst equipped vehicles not tested with petrohol
[C] LTIS subset (11 vehicles) of sample vehicle fleet

The impact of ethanol on the relative contributions of CH_4 and N_2O were minor (< 0.5% difference), compared to that of CO_2. The relative contributions of CH_4 to CO_2-e were small for all vehicle groups (< 1%). The N_2O contribution to CO_2-e was approximately 1.4% for ULP vehicles but was significantly higher for catalyst-equipped vehicles (approximately 8.3%) based on the US estimates.

The contribution of "indirect" greenhouse effects of other gases emitted (such as CO, NO_x, and VOCs which can influence the concentrations of radiatively active GHG) to CO_2-e calculations is not included due to the complex nature of the atmospheric chemical processes and short atmospheric lifetimes of these species. BTRE (2002) estimates that if these "indirect" effects were taken into account the GHG CO_2-e would be significantly higher in the order of 10-20%.

Net CO_2-equivalents From Ethanol Combustion

The term "carbon dioxide equivalent" (CO_2-e) is used as a measure of the total GHG effect and is calculated by applying the GWP shown in Table 14.

According to the GHG accounting rules in the Kyoto Protocol, the CO_2-e of the renewable energy component of transport fuels can be removed from the total fuel CO_2-e due to the atmospheric CO_2 sequestered during crop growth.

APACE (2003a) calculated the CO_2-e contribution from ethanol combustion by subtracting the CO_2-e of petrol fraction (based on base fuel-ULP or LP) from the CO_2-e of the E10 fuel. The CO_2-e of the petrol fraction is calculated by multiplying the CO_2-e calculated for the base fuel by the mass fraction of base fuel in E10 and the ratio of mass fuel consumptions of E10 to the base fuel as represented by:

$$(CO_2\text{-e})_{EtOH} = (CO_2\text{-e})_{E10} - ((CO_2\text{-e})_{BF} \text{ x } m_{BF} \text{ x } M_{E10}/M_{BF})$$

where:

$(CO_2\text{-e})_{E10}$ represents the CO_2-e equivalent emission of E10 (g/km);
$(CO_2\text{-e})_{BF}$ - CO_2-e equivalent emission of the base fuel (ULP or LP) (g/km);
m_{BF} - mass fraction of the base fuel in E10;
M_{E10} - mass of E10 consumed (g/km);
M_{BF} - mass of base fuel consumed (g/km)

The masses of E10 and base fuel consumed (g/km) were converted from the calculated city fuel consumptions (L/km) using the density measurements (at 20°C) of the respective fuels used in the NSW EPA study. The mass fraction of the base fuels in the respective E10 blends were also calculated from the same density measurements (at 20°C). The calculated net CO_2-e emissions are shown in Table 18.

Table 18. Net CO_2-e tailpipe emissions factors (g/km) from Pre- and Post-1986 classed vehicles with petrol and petrohol

	Gross CO_2-e	CO_2-e (Ethanol)[a]	Net CO_2-e[a]
PRE-1986 Petrol	261.4	-	261.4
PRE-1986 Petrohol	265.2	25.7	239.5
POST-1986 Petrol	289.6	-	289.6
POST-1986 Petrohol	291.3	18.6	272.8

[a] Using city cycle fuel consumption

8.3.2.3 Regulated Pollutants (CO/NO_x/THC)

Regulated exhaust emissions in the NSW EPA Petrohol study were conducted in accordance with the ADR37/00. Tailpipe emissions results of the regulated air pollutants (CO_2 included for convenience) from the NSW EPA study are shown in Table 19 for (i) leaded fuelled vehicles (pre-1986 models) and (ii) unleaded fuelled vehicles with a catalytic converter (post-1986 models[14]).

The impacts of ethanol were: significant decrease in CO emissions; reduced HC emissions, and no discernible effect on NO_x for pre-1986 vehicles with a minor increase in NO_x for post-1986 vehicles. Under certain operating conditions, HC and CO formation can be reduced by the addition of oxygenates. The largest improvements have been found with older carburettor vehicles running a rich AFR, relative to stoichiometric operation (API, 2001). This is due to the "leaning effect" from the increased oxygen content of the fuel. The emission control benefit of the oxygenate is greatly reduced in modern vehicles with more enhanced engine management systems using "closed-loop" control (API, 2001; Schifter *et al.*, 2001). Closed-loop control is the automatic fuel delivery regulation in response to oxygen concentration changes detected by an oxygen sensor. Closed-loop control doesn't operate under all conditions; however, with open-loop operation (rich) conditions usually operating during catalyst warm up, idle and high acceleration driving modes.

The results from the NSW EPA study were comparable to those found by AQIRP (refer Section 8.3.2.1), which demonstrated significant reductions of THC (4.9 ± 2.6%), NMHC (5.9 ± 2.7%) and CO but increased NO_x by 5.1% (± 4.1%), using E10 relative to the base fuels.

Table 19. Tailpipe emissions (g/km) and percentage change of regulated air pollutants from Pre- and Post-1986 vehicles using petrol and petrohol

	THC	CO	NO_x	CO_2
PRE-1986 Petrol	1.86	19.13	1.84	255.1
PRE-1986 Petrohol	1.66 (-10.8%)	12.06 (-37.0%)	1.83 (-0.5%)	259.9 (+1.9%
POST-1986 Petrol	0.66	8.45	1.39	264.4
POST-1986 Petrohol	0.57 (-13.6%)	6.18 (-26.9%)	1.46 (+5.0%)	266.2 (+0.7%)

It should be noted that there is an uncertainty concerning the effect of any ethanol (and other potential oxygenated products of combustion), present in the exhaust gas, on the measurement of THC when using the flame ionisation detector (FID) method, which is usually employed for these measurements. The FID responds to all carbon containing compounds, but oxygenated compounds have a lower per-carbon response than hydrocarbons. There would, however, not be expected to be significant levels of unburned ethanol in the tailpipe emissions (but could be more significant with evaporative emissions).

8.3.2.4 Aldehydes and Air Toxics

Aldehyde and air toxics emissions in the NSW EPA Petrohol study were collected during the three phases (Bag 1, 2, 3) of the ADR37/00 emissions tests. The exhaust emissions samples were analysed for approximately 50 species (C_1 to C_{10} hydrocarbons and oxygenated hydrocarbons) by CSIRO–Division of Energy Technology (formerly Division of Coal and Energy). Aldehyde samples were collected from each vehicle but hydrocarbon speciation was conducted only on LTIS vehicles. The study was conducted in two stages: vehicles were tested in the tuned condition with the corresponding conventional petrol as well as with the corresponding petrohol fuel (stage1), and after a period of approximately 12 months the vehicles were retested in both pre-tune and post-tune modes (stage2).

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[14] "Post-1986" includes 1986 year model vehicles.

The tailpipe emissions of aldehydes and air toxics from the NSW EPA study are shown in Table 20 for (i) leaded fuelled vehicles (pre-1986 models) and (ii) unleaded fuelled vehicles with a catalytic converter (post-1986 models). The impacts of petrohol use were a significant increase in mean aldehyde exhaust emissions: formaldehyde (27 ± 12%) and acetaldehyde (215 ± 58%), and a seemingly uniform reduction in aromatics: benzene (23 ± 12%), toluene (21 ± 13%) and xylene (20 ± 12%). Increased acrolein (also known as acrylaldehyde or acrylic aldehyde) and decreased 1,3-butadiene emissions were not statistically significant at the 95% confidence level. Warner-Selph and Harvey (1990) also showed E10 had no discernible effect on 1,3-butadiene tailpipe emissions. There is, however, general agreement that E10 should decrease 1,3-butadiene emissions (Black, 1991; Beer *et al.*, 2001).

The E10 tailpipe results found in the NSW EPA study were comparable to those found in the US AQIRP program (refer Section 8.3.2.1): 13% decrease in benzene; 6% decrease in 1,3-butadiene; 19% increase in formaldehyde, and 159% increase in acetaldehyde. These were further supported by the Warner-Selph and Harvey (1990) study, which found significantly decreased emissions of aromatic compounds from E10 (20 to 60%). Also Guerrieri *et al.* (1995) reported significant acetaldehyde increases with E10 but found formaldehyde emission increases were not statistically significant and this was supported by Warner-Selph and Harvey (1990).

Correlations between air toxics and THC exhaust emissions were found to be essentially the same for petrohol and petrol. The volumetric displacement of 10% (v/v) of the aromatic containing petrol component with ethanol could explain, at least partly, these observed aromatic emissions reductions (as unburned HC).

CSIRO also analysed air toxic (4 compounds) in evaporative emissions of a very small subset of the vehicles (4) but no valid conclusions could be drawn and were omitted from this discussion.

Table 20. Tailpipe emissions (mg/km) of aldehydes and air toxics (and calculated OFP) from Pre- and Post-1986 vehicles with petrol and petrohol

	Form-aldehyde	Acet-aldehyde	Acro-lein	1,3-Butadiene	Benzene	Toluene	Xylene	NMVOC[a]	OFP[b]
PRE-1986 Petrol	31.85	7.58	3.493	18.60	64.83	112.71	86.43	1694	7341
PRE-1986 Petrohol	40.38	24.04	4.017[c]	14.02 [c]	53.30	93.02	71.77	1352	5975
POST-1986 Petrol	5.64	2.24	1.138	1.35	14.36	19.21	16.17	339	1341
POST-1986 Petrohol	7.16	7.00	1.227[c]	1.23 [c]	10.35	14.17	12.29	266	1037

[a] Non Methane Organic Compounds (NMVOC) calculated from total of approximately 50 compounds (HC and oxygenated organics)

[b] Ozone Forming Potential (OFP)

[c] Increase relative to base fuel not statistically significant at 95% confidence level

Total ozone reactivities (mg O_3/mg NMVOC) and total Ozone Forming Potential (OFP) (mg O_3/km) were calculated in the Petrohol study using the Maximum Incremental Reactivity (MIR) method for determining ozone impacts and is discussed in Section 8.3.2.6.

8.3.2.5 Particulate Matter (PM)

Vehicles emit particulate matter (PM) in the exhaust as well as larger particles through mechanical wear of tyres and entrainment of roadside dust. It is likely that both incomplete fuel combustion and engine oil consumption contribute significantly to PM tailpipe emissions. The PM emissions from light-duty passenger vehicles are predominantly (about 85%) composed of carbonaceous material (Cadle *et al.*, 1999). The composition and particle size distributions aspects of PM have been discussed in the Australian context (NEPC, 2001). Very few studies of passenger vehicle emissions of particulate matter have been conducted in Australia. An Alaskan study found that PM10 emission rates decreased when the test vehicles were operated with E10 relative to base gasoline (Mulawa *et al.*, 1997a).

A US EPA study (US EPA, 1993) indicated that PM10 emission rates, from in-use light duty passenger vehicles, tended to increase relative with HC emissions and estimated that PM10 was 1.1% of HC emissions. A similar result was found in the Alaskan study (Mulawa *et al.*, 1997a), with average PM10 emissions at 1.2% and 1.4% of HC emission concentration levels for gasoline and E10 respectively. The results of the FTP emission testing in the Alaskan study showed that most (more than 80%) of the PM10 emissions occurred in the cold start phase (Phase1 or Bag1) of the test sequence and were assumed to be due to enrichment.

US studies have reported that PM10 emission rates from properly maintained, in-use US production vehicles (model years 1986 to 1990) are less than 0.006g/km (Siegl *et al.*, 1994; Mulawa *et al.*, 1997b). A US 24 vehicle emission study found average PM10 emission factors (FTP cycle) of 0.004 and 0.002 g/km for Tier 0 and Tier 1 vehicles, respectively (Cadle *et al.*, 2001). A US study claims that, for properly functioning late model US vehicles, PM10 emission rates are less than 0.001g/km (Chase *et al.*, 2000). Peugeot claims current European technology engines reduce PM10 to 0.002 g/km. It could be expected that the average vehicle fleet PM10 emission rate be in the order of 0.002 g/km by 2010. Current in-use vehicles have PM10 emissions of approximately 0.009g/km. The US EPA Mobile5 emissions model estimate a PM10 emission factor of 0.00932 g/km for petrol passenger vehicles on arterial roads. Elevated emission rates have been found for some poorly maintained vehicles (Cadle *et al.*, 2001; Sagabiel *et al.*, 1997, and Mulawa *et al.*, 1997b).

8.3.2.6 E10 Effects on Ozone Formation

The effects of E10 on ozone formation are ambiguous. The major problem with predicting or quantifying ozone formation is the location specific nature of it, being strongly dependent on: meteorology; topography; chemical composition and mass of the emissions, and the presence and sources of ozone precursors.

The CSIRO component of the Petrohol Study (APACE, 1998) conducted HC speciation of exhaust samples from 11 vehicles and estimated the effects of E10 on OFP and found a decrease of approximately 20%, relative to conventional petrol (refer Table 20). This effect was attributed largely to lower mass emissions of NMVOC. There was little variation in reactivities (mg O_3/mg NMVOC emitted) between petrol and petrohol fuelled vehicles, for either the pre- or post-1986 vehicles. This resulted in a reactivity adjustment factor (RAF) for petrohol close to 1 for both vehicle groups. For petrohol vehicles lower mass emissions of ozone per km were found. This was due to lower mass emissions of NMVOC (approximately 20% lower) and not due to significant reductions in exhaust reactivity. The OFP was calculated using the Maximum Incremental Reactivity (MIR) scale — incremental reactivities would be expected to give good approximations to effects on ozone with the introduction of petrohol (Chang and Rudy, 1990).

The OFP results presented here should be treated with caution, as the OFP is a function of the characteristic of a particular air shed. In urban areas large reservoirs of NO_x can exist, which can react with VOCs to form ozone. If the VOC concentrations are small, the amount of ozone produced would also be small. In the absence of NO_x even large VOC concentrations would not be expected to produce O_3.

The effects of ethanol in petrol on ozone formation have been extensively studied in the USA, with respect to replacing the MTBE oxygenate. This limits the relevance of these studies to Australia but qualitative information can be utilised. A study compared the emissions using an E10 fuel (high RVP) to an MTBE containing fuel (normal RVP) and found that OFP increased by 17% (CARB, 1998), primarily as a result of a 40% increase in evaporative emissions with E10.

Many US studies have concluded that the overall OFP of ethanol fuel blends is the same or lower than that of petrol (American Coalition for Ethanol website). Other overseas studies (NRC, 1999) are ambiguous as to whether the addition of 10% ethanol is beneficial or is detrimental to air quality in terms of ozone. Howard (in NSTC, 1997) concluded that gasoline with ethanol would result in adverse ozone impacts associated with increased NO_x and VOC emissions.

The effect of the evaporative emissions (on OFP) from E10 is discussed in Section 10.4.1.

8.3.3 Evaporative Emissions

Evaporative emissions from petrol fuelled motor vehicles can form a significant proportion of total downstream vehicle HC emissions, which can effect ozone formation and the levels of air toxics. Due to the low volatility or RVP of diesel fuels, evaporative emissions are not as significant as those from petrol fuelled vehicles.

8.3.3.1 Background

Evaporative emissions are fuel (and oil) vapours that are released by the vehicle prior to combustion. The factors that effect evaporative vehicle emissions are: fuel volatility, ambient temperature, driving conditions, as well as vehicle specific factors such as vehicle design and maintenance, and type and condition of evaporative emission control systems (if fitted to the vehicle) (Environmental Protection Authority of Victoria, 1997). Evaporative hydrocarbons emissions from vehicles are a result of:

- diurnal losses (ambient air temperature affects fuel temperature giving vapour loss;
- running losses (vaporisation of fuel while vehicle is running and generating heat);
- hot soak losses (vapour loss after vehicle stops and cools down);
- resting losses (fuel permeation through rubber engine components and liquid fuel-leakage); and
- refueling losses (vehicle refuelling and bulk tanker refilling)

With the exception of resting losses, all of the sources of evaporative emissions increase as a function of increasing fuel volatility and ambient temperature. The determination of vehicle evaporative emissions from running losses is difficult to do and have never been studied in Australia.

Vehicles must comply with the relevant ADR for regulated evaporative HC emissions via a “SHED” test. This test involves summing the emissions recorded from two tests: “Diurnal Soak” and a “Hot Soak”.

The US EPA has introduced upgraded evaporative emissions certification test procedures, under the Tier 2 regulations, replacing the standard SHED test by the "Multiday Diurnal" SHED test method, considered by US EPA to be a better model for "real world" conditions.

8.3.3.2 Petrohol (E10)

The addition of ethanol to petrol increases the fuel blend volatility (refer Section 8.1.2), which leads to increased evaporative emissions.

Limited studies exist, particularly in Australia, which have quantified evaporative vehicle emissions from conventional and ethanol-blend fuelled vehicles. US studies have, however, shown that higher evaporative emissions occur as a result of ethanol blended fuels, relative to petrol and that these resulted from the higher vapour pressure and differences in distillation characteristics of the fuel (API, 2001).

The NSW EPA Petrohol study conducted regulated evaporative emission tests (SHED tests according to ADR37/00). The emissions for pre-1986 (17 tests) and post-1986 (39 tests) vehicles were tested using ULP and petrohol and the results are shown in Table 21. Large increases in evaporative emissions (both diurnal and hot soak) were found when using petrohol relative to ULP, of the order 33% and 21.8% for pre-1986 and post-1986 vehicles respectively. The high evaporative emissions from the E10 fuel were attributed to the elevated RVP of the blended fuels (about 5 kPa increase). These results are comparable to those found for E10 by the California EPA (CARB, 1998), which showed an RVP increase from 48 kPa to 55 kPa, and subsequent increase in the evaporative HC emissions by an estimated 40%, relative to conventional gasoline. And similarly, an earlier US study (Warner-Selph and Harvey, 1990) found an approximately 30% increase in evaporative emissions from evaporative emission tests conducted on 5 cars (wide range of emission control systems) using an E10 fuel.

Table 21. Regulated evaporative emissions (g/test) from pre- and post-1986 vehicles with petrol and petrohol

	Mean Total Evaporative Emissions (g/test)	Evaporative Emission Increase- using Petrohol (%)	RVP (kPa)
PRE-1986 Petrol	19.54		82.3
PRE-1986 Petrohol	25.98	33 (±17)	87.4
POST-1986 Petrol	10.54		80.6
POST-1986 Petrohol	12.84	22 (±17)	85.4

APACE re-assessed the NSW EPA results and removed six outliers for post-1986 vehicles ("1986-on") and four outliers for pre-1986 and the findings, including results from linear regression analysis, are shown in Table 22 (APACE, 1998). The results show higher hot soak evaporative emissions. In addition, this table shows that E10 had a significantly larger impact on hot soak emissions, relative to diurnal emissions, and shows the large uncertainties present in these findings.

The NSW EPA study suggested that diurnal evaporative emissions were influenced directly by fuel vapour pressure whereas hot soak emissions were related to the distillation characteristics of the blend.

Table 22. Effect of petrohol on regulated evaporative emissions from pre- and post-1986 vehicles (outliers removed by APACE)

Vehicle group	Original NSW EPA key findings (% incr. evap. emissions with E10)	Outliers removed key findings (% incr. evap. emissions with E10)	APACE linear regression (outliers removed) (% incr. evap. emissions with E10)	
Diurnal				
PRE-1986	15 (±15)	17 (±15)	14%	R^2=0.97
POST-1986	17 (±16)	17 (±15)	9%	R^2=0.89
Hot soak				
PRE-1986	51 (±30)	58 (±29)	60%	R^2=0.92
POST-1986	34 (±26)	42 (±31)	33%	R^2=0.85

Source APACE, 1998 (Table 5-22, p. 47)

It should be noted that there is an uncertainty concerning the effect of any ethanol (and other potential oxygenated products of combustion), present in the evaporative emissions, on the measurement of HC when using the flame ionisation detector (FID) method, which is usually employed for these measurements. The FID responds to all carbon containing compounds, but oxygenated compounds have a lower per-carbon response than hydrocarbons. With E10, ethanol could be present in significant concentrations in the evaporative emissions, which could underestimate the HC evaporative emissions.

8.3.3.3 Other Issues

A "Co-mingling" effect could be important if E10 is not universally supplied in the fuel market place, as increases in vapour pressure could result if fuels with different ethanol concentrations were mixed in petrol tanks (and storage tanks). Dilution of the fuel ethanol content from 10% could result in increases in vapour pressure (maximum RVP at approximately 3-5% ethanol).

Any steps, if required, to reduce the evaporative emissions from E10 use would require processes to alter the fuel RVP at the refinery level. Other steps to contain evaporative emissions could be employed such as to improve the effectiveness of in-vehicle evaporative emissions control systems.

Evaporative Emission Control Technologies

Evaporative emission system technologies designed to reduce evaporative emissions from vehicles using gasoline and E10 have also been examined (CEC, 1999). With E10, evaporative emissions were almost twice as high as when using gasoline without ethanol due to much higher permeation rates. This study also showed that by using low permeation materials, evaporative emissions could be substantially reduced from both fuels.

Carbon canisters are the primary components of the evaporative vehicle emission control strategies. A carbon canister acts by trapping vapours on the activated carbon surfaces and subsequently releasing them during vehicle operation (vapour is directed to intake manifold). If the canister capacity is exceeded the excess vapours are vented to the atmosphere. The working capacity of carbon canisters could potentially be reduced by ethanol due to:

- oxygenated compounds binding more tightly to the carbon surfaces than HC (Furey and King, 1980; Grisanti *et al.*, 1995; Duncan, 2002); and
- high water affinity of ethanol causing water degradation to canister capacity (Manos *et al.*, 1977).

Real-World Evaporative Emissions

The Australian National Vehicle in-Service study (NISE) (FORS, 1996) found that "real-world" evaporative emissions were many times greater than the maximum ADR specifications. This was not limited to pre-1986 vehicles (more than 80% exceeded ADR limit) as the average evaporative emissions of the post-1986 vehicles tested were four times the ADR limit (more than 80% exceeded ADR limit). This was consistent with the NSW study where 88% and 74% of the pre-1986 and post-1986 vehicles, respectively, exceeded the ADR limits. A significant cause of this was believed to be the difference in vapour pressure of the commercial grade fuel, which was used in the tests, relative to the ADR certification fuel (ADR fuel has a lower vapour pressure and lower mid-range distillation characteristics).

Results using evaporative testing standards are given as mass of evaporative emissions per test (g/test) and it is, therefore, difficult to assess the relative contribution of evaporative HC emissions to the total downstream vehicle HC emissions. Environment Australia does, however, estimate evaporative emissions on a per kilometre basis in the *National Pollution Inventory* (Environment Australia, 2000b), using the "estimated emissions technique". This method uses the US EPA emissions model MOBILE5a and included typical average figures for RVP, temperature and number of vehicle trips per day. Refuelling evaporative losses were not included in this estimation as they were covered in the *NPI Manual* for *Aggregated Emissions from Service Stations*. Using this method, the evaporative emission factor estimated for petrol passenger vehicles was 0.535 g/km based on arterial road usage, an average daily temperature of 15 °C (minimum of 10 °C and maximum of 20 °C) and a RVP of 10 psi (69 kPa). The estimates were not segregated into pre-1986 and post-1986 vehicles. This estimate would indicate that evaporative vehicle HC emissions were about the same order of magnitude as tailpipe HC emissions, according to the exhaust HC results from the Petrohol study.

Speciation of Evaporative HC Emissions

The addition of ethanol to petrol has the effect of decreasing the boiling point of the hydrocarbon species in the petrol, with a slightly smaller impact observed for aromatic compounds (API, 2001). In a BTCE report the composition of the evaporative emissions was concluded to be the same for ethanol blends as gasoline (BTCE, 1994). The impact of ethanol would not significantly effect the relative concentrations of hydrocarbon species in the evaporative emissions (Grisanti *et al.*, 1995).

As mentioned already (Section 8.3.2.4), as a component of the NSW EPA Petrohol study, CSIRO also analysed air toxics in evaporative emissions of a very small subset of the vehicles (four) but no valid conclusions could be drawn and were omitted from the discussion.

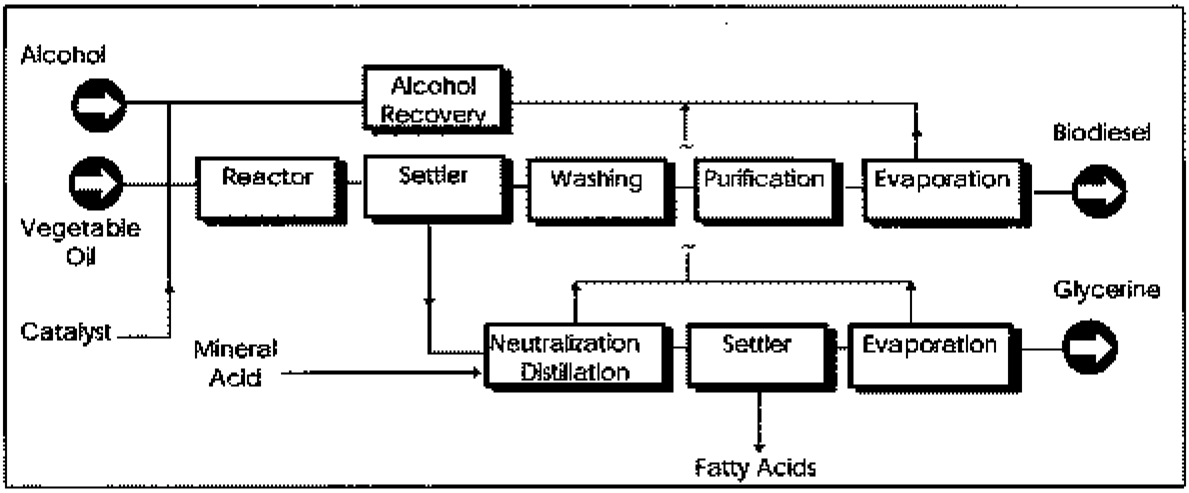
In addition, the Orbital (2003) study, assessing the impacts of 20% ethanol in petrol, found little or no detectable CH_4 contribution to evaporative emissions based on the "Hot soak" test. Evaporative emissions, therefore, would be expected to impact on air toxics and ozone forming potential issues without significant GHG considerations. The impact of the evaporative emissions, in terms of the level of air toxics and ozone forming potential, is discussed in Section 10.4.1.

9 EMISSIONS FROM BIODIESEL USE

9.1 Background

Biodiesel is a generic name for fuels obtained by transesterification of a vegetable oil. This produces a fuel with very similar combustion properties to pure diesel, but with lower viscosity. Often biodiesel refers to rapeseed oil methylester (RME), the main European biodiesel. Esterified soybean oil is the main United States source of such fuel, called Soy diesel. Figure 7 depicts a biodiesel production flow chart involving the esterification process.

Figure 7. Flowchart of the process of esterification to create biodiesel fuel



Source National Biodiesel Board production factsheet

Biodiesel can be used in a diesel engine without modification. Mittelbach (1998) quotes a cetane number of 48 for rapeseed methyl ester. Biodiesel has an energy content of 33.3 MJ/L, which is lower than diesel with an energy content of 38.6 MJ/L. This leads to increased fuel consumption when pure biodiesel is used (Taberski *et al.*, 1999).

The greenhouse gas emissions arising from the process depicted in Figure 7 depend on the amount of fossil fuel involved in the production of the alcohol. Even though biodiesel can be made from either ethanol or methanol, the use of methanol simplifies the production process. If methanol is used then this process is described by the equation:

$$\underset{\text{(Triglyceride)}}{C_3H_5(OOCR)_3} + \underset{\text{(Methanol)}}{3CH_3OH} \rightarrow \underset{\text{(Methylester)}}{3RCOOCH_3} + \underset{\text{(Glycerine)}}{C_3H_5(OH)_3}$$

The term “triglyceride” in the equation may be either vegetable oil or tallow. From a chemical point of view, the differences between various plant and animal derived fats are due to the structural variations of fatty acids contained in fat molecules.

The greenhouse gas emissions arising from the process depicted in Figure 7 depend mostly on the amount of fossil fuel involved in the production of the alcohol as given by Sheehan *et al.* (1998a: p. 147), who estimate that 5% (by mass) of the carbon emissions are fossil-fuel carbon. For example, if methanol is used, overall emissions will be higher because current production of methanol involves solely fossil-fuel feedstocks such as natural gas or coal. By contrast, if the use of ethanol produced from renewable resources (biomass) using bioprocesses is contemplated, greenhouse emissions will be lower. Methanol can be produced by the gasification of biomass but this is currently not done.

9.1.1 Whole Grain Oil Seeds and Canola Oil

A considerable number of new project proposals are based on the utilisation of whole grain oilseeds, and canola in particular. However, as internationally traded agricultural commodities, oilseed prices vary considerably depending on both domestic market conditions (i.e. drought) and international market developments. ABARE’s current medium term forecast for canola is for the price to ease considerably over the medium term from its current high level (refer Figure 5) to average $353 a tonne (in real terms) in 2007-2008 (Connell *et al.,* 2003). As with ethanol produced from whole grains, crushed grain meal is also a valuable co-product in the production of biodiesel from whole grains. In this analysis it is assumed canola grain meal is priced at $140 a tonne providing a revenue credit of 36c/L of biodiesel produced.

Based on these figures the net revenue required to cover costs is estimated to be 76c/L. In the case where the raw feedstock is canola oil rather than whole seeds the costs of production are significantly higher ($1.19 per litre) reflecting both the higher cost of the feedstock and the lack of a grain meal co-product.

9.2 Biodiesel Upstream Emissions

The literature examining upstream emissions associated with biodiesel production has been reviewed in Appendix II and will not be discussed in detail here.

9.2.1 Canola Biodiesel

This scenario is taken largely from the data in the CSIRO *Comparison of Transport Fuels* study (Beer *et al*., 2001) with a number of modifications, such as:

- an estimate is made of the environmental impacts from capital construction;
- a credit is allowed for the production of glycerine (rather than an economic allocation);
- a credit is allowed for the production of canola meal (rather than an economic allocation); and
- transport emissions for distribution of biodiesel are included.

9.2.1.1 Glycerine and Canola Meal Allocation

Glycerine is a co-product in biodiesel production, from the transesterification of oil, and is also produced via other processes involving vegetable oils. Relatively small volumes of glycerine are produced via the synthesis of epichlohydrin from allylchloride and propylene, and via the fermentation of sugar and other starches. Table 23 shows the different production sources and the volumes produced of glycerine since 1992. Glycerine production from biodiesel has increased steadily from 1992 but is still a relatively small contributor (less than 7%). Glycerine competes in the marketplace with other polyols, a range of which is shown with their respective prices in Table 24. Propylene glycol has been used as an equivalent substitute for glycerine, as it is in the middle range for price competitiveness, and because production data are available for use in the LCA.

Table 23. Glycerine production by source in Gt

	1992[1]	1995 [a]	1998 [b]
1. Soap glycerine	208	210	199
2. Hydrolyser glycerine	268	286	304
3. Ester glycerine			
i) ex fatty alcohols	78	99	109
ii) ex biodiesel	6	40	55
iii) ex fat substitutes		-	3
iv) other esters		15	20
4. Synthetic glycerine	78	80	95
5. Fermentation		-	15
Total	638	730	800

Sources [a] World Glycerine Conference 1992 and 1995
[b] HBI Estimates[15]

Canola meal is commonly used in stockfeed for energy and particularly protein. Crops that are grown specifically for fodder (and for soil nitrogen replacement) are lupins and field peas. Lupins have a protein content of around 35% and energy content around 19%. This is almost identical to the protein and energy content in canola meal (36% protein and 20% energy content) so canola meal is assumed to displace lupins on a 1:1 basis.

Table 24. Prices of competing polyols (December 1995)

Polyol	US$/ton
Glycerol	1,555
Trimethylolpropane	1,278
Pentaerythritol	1,278
Propylene glycol	1,000
Glycerol (Mar. 98)	833
Ethylene glycol	667
Sorbitol (70%)	400

Source HBI.
Commodity prices are based on the US short ton

9.2.2 Transport Emissions for Distribution of Biodiesel

Using the same assumptions as described in the distribution of ethanol from wheat production, the total additional road transport for biodiesel is shown in Table 25 as 303 tonne km per kL, or 344 tonne km per tonne of biodiesel. This would be offset by the shipping of the 19% of diesel used in Australia.

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[15] http://www.hbint.com/glycerine.htm

Table 25. Transport assumption for 350 ML of biodiesel from canola oil

State	Biodiesel volume[1] (ML)	Distance of major centre from biodiesel production (km)	Road transport (ktonne km)
Vic	116.7	400	41,067
NSW	135.3	400	47,637
Queensland	0	0	
SA	37.3	200	6,571
WA	60.7	200	10,677
Tas	0	0	
NT	0	1,800	
Total	350		105,952

[1] Production in each State is based on average petroleum consumption by State. Major canola producing States only included.

9.2.3 Tallow Biodiesel

This scenario is taken largely from the same data developed in the CSIRO *Comparison of Transport Fuels* study (Beer *et al.*, 2001) with a number of modifications:

- estimate of the environmental impacts from capital construction and a credit for the production of glycerine – as described in the previous section for canola based biodiesel; and
- reduced transport emission from diesel distribution, with biodiesel from tallow assumed to be produced in the major cities, therefore eliminating the need for additional transport.

Though there is some debate about the service station infrastructure needed to distribute biodiesel, no additional infrastructure is assumed in this assessment.

9.3 Downstream Emissions

Significant uncertainty is associated with the determination of tailpipe emissions, varying according to engine technology and condition, vehicle maintenance and also, if non-steady state test cycles are used, the accuracy with which the cycles have been performed by the test driver. Emission testing has been conducted with a wide range of vehicles utilising a range of pollution control strategies. It is also difficult to compare the combustion emissions of substantially different fuels such as biodiesel and LSD.

Concerning the emissions from diesel vehicles, there are some generalisations associated with different fuels, which include:

- the less volatile and more aromatic the fuel, the higher the exhaust PM emissions;
- presence of sulfur in the fuel results in increased PM; and
- oxygenated fuels reduce PM due to more complete combustion, assuming that other fuel qualities (e.g. cetane number) remain constant.

In regard to fuel consumption, provided the fuel is within the normal specification range, then for a given engine technology and transport task, fuel economy will be related to the energy content of the fuel. The higher oxygen content and accompanied lower calorific content of biodiesel, relative to diesel, results in increased fuel consumption.

The extensive use of biodiesel fuels in the United States and Europe means that data is available on their emission characteristics during operational performance and this data was summarised by Beer *et al.* (2000). In Beer *et al.* (2001), more recent results, and some of the relevant older results were reviewed and the different studies were compared. US tailpipe data is presented on the Alternative Fuels Data Centre (AFDC) website and is covered in Beer *et al.* (2000).

There were discrepancies found between European and American biodiesel emissions results (Beer *et al.*, 2000, 2001). European data (Arcoumanis, 2000) suggested biodiesel gives a reduction in HC compared with LSD. CO tends to be lower for biodiesel, while NO_x tends to be slightly higher. Biodiesel emissions of PM have been reported to be lower (US EPA, 2002; Sharp, 1998; Buckmann and van Malsen, 1997) or it may be higher (Arcoumanis, 2000; Ceuterick and Spirinckx, 1999) relative to conventional diesel.

For heavy vehicles, the estimates of greenhouse gas and regulated air pollutants are based on four main studies: Beer *et al.* (2001), Graboski *et al.* (1999), Sharp (1998), and US EPA (2002). A review of biodiesel tailpipe emission studies is given in Appendix I.

9.3.1 Light Vehicle Use

The estimated downstream emissions (g/MJ) of CO_2 and regulated pollutants for BD5, BD20 and BD100 fuels using light duty 4WD vehicles are shown in Table 26. Several studies have been reviewed (Appendix I) and the tailpipe emissions results used for this study were derived from a compilation of these sources.

Table 26. Downstream emissions of CO_2 and regulated pollutants

Type vehicle	Fuel		CO_2 (g/MJ)	HC (g/MJ)	CO (g/MJ)	NO_x (g/MJ)	PM (g/MJ)
4WD	BD100	**Best estimate**	**0**	**0.038**	**0.18**	**1.14**	**0.023**
		Min		0.02	0.14	0.83	0.023
		Max		0.04	0.184	1.156	0.033
	BD20	**Best estimate**	**66**	**0.05**	**0.25**	**1.05**	**0.03**
		Min	63.6	0.05	0.2	0.925	0.03
		Max	66.7	0.08	0.4	1.05	0.041
	BD5	**Best estimate**	**66.5**	**0.06**	**0.3**	**1.00**	**0.033**
		Min	64	0.06	0.22	0.834	0.033
		Max	66.8	0.082	0.4	1.05	0.036

The main findings from the quantitative studies are presented below:

1. In the UK, a study conducted by Groves (2002) showed the variability of the well-to-wheel (WTW) greenhouse gas emissions from canola biodiesel. The emissions on a per km basis varied between production scenarios: 90 g/km (for canola imported from Australia) and 207 g/km (from conversion of land). The derivation is based on the assumption that canola and rapeseed biodiesel have identical combustion properties. Groves used a 1.8L Ford Focus reference vehicle with a vehicle energy of 1.91 MJ/km and WTW energy of 2.04 MJ/km. The diesel (< 50 ppm sulfur) emissions were 157 g/km (141 in tailpipe and 16 in upstream) and the 5% blend resulted in 152.8 g/km of GHG emissions.
2. Similar results are provided in the GM-LBST (2002) study, conducted in Europe. The report – a replica of the GM study conducted in the US one year earlier — indicated that on a TTW basis, a 5% biodiesel blend (BD5) results in GHG emissions of between 138 to 157 g/km (best estimate = 146 g/km), using a 2002 Opel Zafira minivan (1.8L, 5-speed and MTA).
3. The American GM study determined the following tailpipe greenhouse gas emissions: for 5% biodiesel LDV, TTW GHG emissions are between 138 to 145 g/km (best estimate =139

g/km) can be used. For 20% biodiesel LDV the TTW GHG emissions would vary between 131 and 153 g/km (best estimate =132 g/km).

4. With respect to the air pollutants, the main reference source used in this study is the Sharp (1998) report, which presented tailpipe emissions results from tests run on a 1995 Cummins B5.9 (full-size pick-up).

9.3.2 Heavy Vehicle (Truck and Bus) Use

For heavy vehicles, the estimates of greenhouse gas and regulated air pollutants are based on four main studies: Beer *et al.* (2001), Graboski *et al.* (1999), Sharp (1998), and US EPA (2002).

The first three provide comparative values of emissions for different types of vehicles fuelled with diesel and biodiesel from different feedstocks, the last one is based on a statistical analysis of 39 American studies and reports the changes in emissions from use of biodiesel blends compared to diesel.

Beer *et al.* (2001) values of GHG and regulated pollutant emissions for trucks and buses, presented in the *Comparison of Transport Fuels* study, were compared with the values reported in the American studies and "updated" with the more recent data from the US EPA study (2002) as shown in Table 27.

US EPA (2002) indicated that a 5% biodiesel blend leads to CO_2 emissions increase of about 0.25% (for clean base fuel), with no change in the overall average (all fleet). For 20% biodiesel blend, changes in CO_2 emissions varies between -0.35% (for animal-based biodiesel) to +0.4% (for clean based fuels). US EPA (2002) also concluded that the changes in NO_x, PM, and CO emissions, due to biodiesel, were the following:

- for 5% biodiesel: 0.7% increase in NO_x, 3% reduction PM, and 2% reduction CO,
- for 20% biodiesel: 0.5 (animal-based biodiesel) to 2.5% increase in NO_x (soy-based biodiesel), 6.7 (soybean/rapeseed biodiesel) to 10% reduction in PM (animal based biodiesel); and 4 (soybean based biodiesel) to 9% reduction in CO (animal-based biodiesel).

The Sharp (1998) study (mentioned in Section 9.3.1) provided tailpipe emissions from tests run on a 1997 Cummins N14 (highway truck) and a 1997 DDC Series 50 (urban transit bus) on the US EPA heavy-duty transient Federal Test Procedure (FTP). The use of biodiesel had the effect of lower emissions of CO, PM, and HC, with some increases in NO_x emissions on some engines. Biodiesel has also increased the catalytic converter efficiency in reducing PM.

Graboski *et al.* (1999) analysed data on emissions based on the EPA heavy-duty transient cycle for 27 pure biodiesels and three 20% blends. The biodiesel was obtained from vegetable and animal feedstocks. All biodiesel fuels (pure or in blend with diesel) reduced PM (13% for blends up to 50% for neat biodiesel). NO_x emissions increased, but the amount of additional NO_x varied considerably upon the feedstock. CO emissions were 37% (for pure biodiesel) and 2.3% (for blends) lower for biodiesel. In addition, THC decreased as well by about 10%, as compared to diesel and no conclusive results were obtained for CO_2 emissions.

Table 27 shows the emission values (best estimates and ranges) estimated for trucks and buses by Beer *et al.* (2000, 2001) normalised in such a way that the values all refer to the vehicles examined by Beer *et al.* (2001).

The "fleet" values from *Comparison of Transport Fuels* study are within the range of values tested in the US, with only one exception for NO_x, where emissions/MJ are approximately half of those reported in the US study.

Table 27. Tailpipe emission factors from Beer *et al.* (2001)

Type vehicle	Fuel		CO_2 (g/MJ)	HC (g/MJ)	CO (g/MJ)	NO_x (g/MJ)	PM (g/MJ)
Truck	BD100	**Best estimate**	**0**	**0.057**	**0.15**	**1.05**	**0.026**
		Min		.003	0.136	0.8	0.023
		Max		0.06	0.184	1.2	0.029
	BD20	**Best estimate**	**66.7**	**0.07**	**0.22**	**0.96**	**0.031**
		Min	65	0.065	0.21	0.925	0.03
		Max	67	0.075	0.24	0.986	0.04
	BD5	**Best estimate**	**66.8**	**0.07**	**0.23**	**0.95**	**0.035**
		Min	65	0.07	0.22	0.834	0.034
		Max	67	0.082	0.26	0.95	0.036
Bus	BD100	**Best estimate**		**0.038**	**0.2**	**1.13**	**0.025**
		Min		0.03	0.136	0.8	0.02
		Max		0.039	0.3	1.2	0.029
	BD20	**Best estimate**	**66.8**	**0.07**	**0.35**	**1.05**	**0.031**
		Min	65	0.06	0.21	0.925	0.03
		Max	67	0.075	0.35	1.06	0.04
	BD5	**Best estimate**	**66.9**	**0.07**	**0.4**	**1.00**	**0.033**
		Min	65	0.07	0.22	0.834	0.033
		Max	67	0.082	0.4	1.05	0.036

Regarding the effect of biodiesel blends on air toxic emissions, the studies agreed that biodiesel use leads to lower values of emissions of most air toxics (except for example, acrolein – slight increase mentioned by Beer *et al.*, 2001).

Beer *et al.* (2001) concluded that relative to diesel, the biodiesel use leads to:

- substantially decreased emissions of PAH and nitro-PAH compounds (30% with a catalytic converter, 12% without a catalytic converter) and virtually eliminated some of the heavier NPAH compounds;
- substantially reduced emissions of aldehydes and ketones;
- dramatic change in the character of the heavier HC species, with only the esters that made up the biodiesel remaining in exhaust among the higher molecular weight HC.

US EPA (2002) estimated that the percentage change in total air toxics is -0.00158 *% biodiesel in the blend. Test results from Sharp (1998) showed decreased emissions of the air toxics measured as given in Table 28, which are consistent with Graboski *et al.* (1999) results.

Table 28. Air toxics emissions

	VOCs (g)	PAH (µg)	Benzene (mg)	Butadiene (mg)
Biodiesel 20% (/MJ)	0.820	2.051	0.179	0.130
Diesel (/MJ)	1.156	3.169	0.802	0.884

10 ENVIRONMENTAL IMPACTS OF BIOFUELS

10.1 Environmental Performance of Ethanol Compared to Unleaded Petrol (ULP)

There are five atmospheric issues on which it is difficult to make firm statements in relation to the environmental performance of ethanol (used as E10) because past studies have produced conflicting results. These are the:

1. full-fuel cycle greenhouse gas emissions;
2. emissions of methane and nitrous oxide;
3. ozone forming potential of petrohol;
4. emissions of air toxics; and
5. whether the benefits of octane enhancement outweigh the disadvantages of the higher vapour pressure.

The environmental performance of ethanol E10 has been analysed based on the APACE (1998) data set and compared to international emission factors. The "post-1986" vehicle values presented by APACE (1998) have been considered as tailpipe averages for this study. These values were then normalised on the basis of the Australian Urban Drive Cycle (AUDC) protocol applied to passenger vehicles. The "Tank-to-Wheel" (TTW) emission performance has been evaluated for a fuel consumption rate of 4.63 MJ/km.

The variability in the emission rates noticed in the literature review is considered in the sensitivity and uncertainty analysis (Chapter 13).

The general method used in the analysis is the same as the life cycle emissions analysis of alternative fuels for heavy vehicles undertaken by Beer *et al.* (2000, 2001, 2002). The methodology used in the above-mentioned studies follows the international standards in life cycle analysis (LCA).

The results are reported as emissions per kilometre, this metric being the most relevant to the emissions result from use of a new fuel.

10.1.1 Greenhouse Gas Emissions per km

The life cycle inventory presented here considers the major direct contributors to greenhouse gases: CO_2 (the main GHG), CH_4 and N_2O, using the IPCC GWP coefficients (refer Table 14).

Table 29 displays the change (%) in full life cycle GHG emissions (on a per km basis) for E10 (obtained from four different ethanol feedstocks), as compared to ULP (Table 29 also contains the upstream and downstream emission factors (g/km) for ULP). The complete upstream and downstream emission factors are shown in tabulated and graphic form, in Appendix VII.

The ethanol blends require more energy in the upstream processes and result in higher GHG emissions. However, the larger reduction in the combustion of E10 leads to savings between 7 and 21 g GHG emissions/km (or between 1.7 and 5.1%) for wheat and molasses co-generation, respectively.

Figure 8 graphically presents the GHG emissions per km for passenger vehicles with the life cycle (WTW) emissions shown as partitioned into upstream (WTT) and tailpipe (TTW) emissions.

Table 29. Percentage change of full life cycle GHG emissions (as g/km) from E10 relative to ULP (%)

Impact category (%)	**E10 (ULP) (molasses cogen energy)**	**E10 (ULP) (molasses)**	**E10 (ULP) (sorghum)**	**E10 (ULP) (wheat)**	**E10 (ULP) (wheat starch waste)**	***ULP (g/km)***
Upstream						
CO_2	4.7	15.0	23.9	25.6	17.0	*56.21*
CH_4	-9.2	-7.9	-4.0	-4.2	-6.3	*0.54310*
N_2O	1329.8	1359.0	-1152.3	3073.3	46.2	*0.00033*
GHG (CO_2-e)	4.3	13.2	17.4	25.1	13.1	*67.717*
Tailpipe						
CO_2	-7.0	-7.0	-7.0	-7.0	-7.0	*340.50*
CH_4	3.7	3.7	3.7	3.7	3.7	*0.00716*
N_2O	-0.1	-0.1	-0.1	-0.1	-0.1	*0.00227*
GHG (CO_2-e)	-7.0	-7.0	-7.0	-7.0	-7.0	*341.354*
Life cycle						
CO_2	-5.4	-3.9	-2.7	-2.4	-3.6	*396.71*
CH_4	-9.0	-7.8	-3.9	-4.1	-6.2	*0.55026*
N_2O	168.2	171.9	-145.9	388.8	5.8	*0.00260*
GHG (CO_2-e)	**-5.1**	**-3.7**	**-3.0**	**-1.7**	**-3.7**	*409.072*

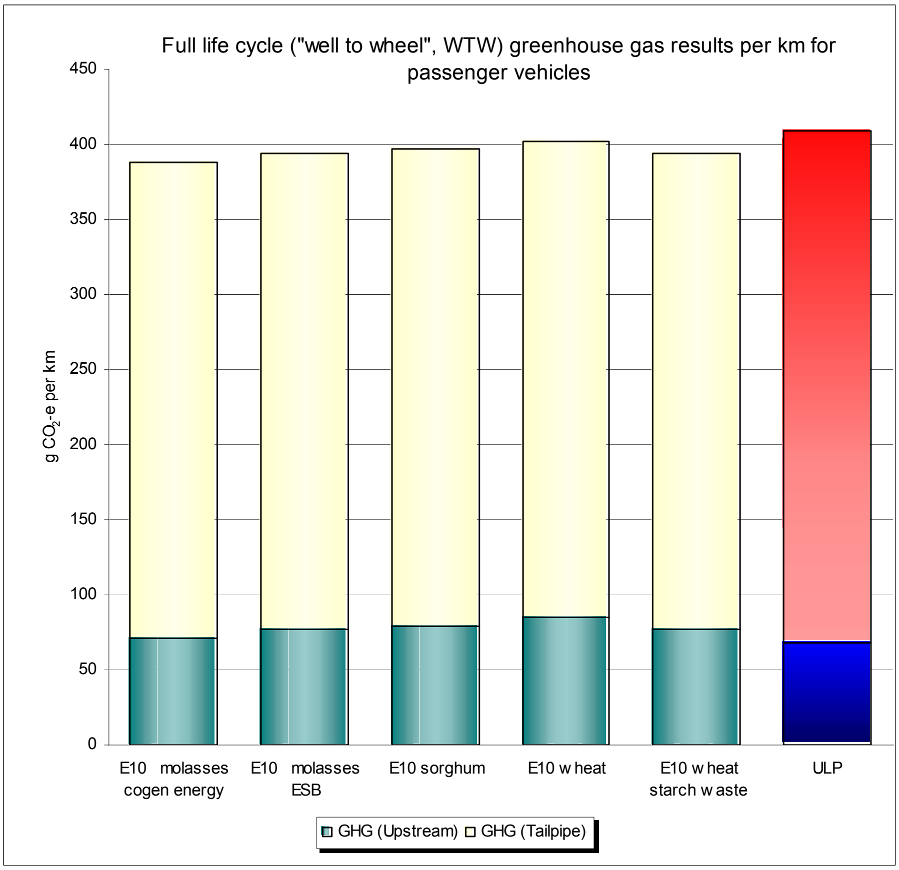

Figure 8. **Life cycle greenhouse gases from E10 and ULP separate for upstream and tailpipe (passenger car)[16]**

10.1.2 Air Pollutant Emissions per km

There are six categories of criteria pollutants associated with transport: particulate matter (PM), oxides of nitrogen (NO_x), carbon monoxide (CO), sulfur dioxide (SO_2), ozone (O_3), and lead (Pb). Of these, ozone is not emitted from tailpipes but is formed from a secondary chemical reaction between NO_x and non-methane volatile organic compounds, NMVOC (NMVOC aggregate many species, including air toxics). The pollutants SO_2 and Pb are no longer considered to be environmental problems in Australian urban airsheds. Thus these three criteria pollutants are not considered any further.

The various criteria pollutants have different health impacts, summarised in the following:

- PM exposure is considered an important risk factor for cardiopulmonary disease and mortality and is associated with other health endpoints, such as chronic bronchitis. Health-based standards are set for PM10 and recently have been introduced for PM2.5[17], because the smaller size particles can penetrate more deeply into the lungs and cause more damage;
- current data and research findings also show evidence of mutagenic and carcinogenic effects of NMVOC. These are believed to arise from the air toxics component of the NMVOC;
- the effects of SO_2 and NO_2 (both known as acid gases) on human health include respiratory problems and damage to the immune system;

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[16] The bottom of the histogram always represents upstream emissions. The ULP bar is coloured differently to emphasise that it is the reference fuel.

[17] www.ephc.gov.au/nepms/air/air-varation.html

- NMVOC and NO_x are also precursors of O_3;
- CO reduces oxygen transport system in blood.

The atmospheric dispersion of these air pollutants depends upon: their own properties (reactivity, weight, particle size); the atmospheric and mixing conditions, and topography. Analysis in this study was restricted to three air pollutants (CO, NO_x, and HC), due to their relevance and data availability.

Table 30 displays the change (%) in full life cycle emissions of the criteria pollutants (on a per km basis) for E10 (obtained from four different ethanol feedstocks), as compared to ULP. The complete emission factor dataset is presented, in tabulated and graphic form, in Appendix VII. In addition, WTW criteria pollutant process tree charts for E10 are also presented in Appendix VII.

Compared to ULP, E10 showed lower CO and NMVOC tailpipe emissions per km, similar PM, and slightly higher NO_x. In the upstream processes, CO and PM emissions are higher for E10 than ULP. NO_x and NMVOC upstream emissions for E10 are either lower or higher than for ULP, depending on the feedstock.

On a full life cycle basis, CO emissions are significantly reduced: 1-1.5 g/km lower, corresponding to a 21-26% reduction for E10 compared to ULP. NO_x is 0.1-0.6 g/km higher for E10, which corresponds to an increase of 2 to 9%. NMVOC emissions are generally lower for E10 (except for wheat) and PM emissions are increased, being 6-7 g/km higher (except for E10 sourced from molasses using co-generation energy).

Figure 9 graphically presents the full life cycle (WTW) CO, NO_x, and NMVOC emissions (g/km) for E10 and ULP (passenger car).

Table 30. Percentage change of full life cycle air pollutants emissions (as g/km) from E10 and ULP (passenger car) (%)

Impact category (%)	**E10 (ULP) (molasses cogen energy)**	**E10 (ULP) (molasses)**	**E10 (ULP) (sorghum)**	**E10 (ULP) (wheat)**	**E10 (ULP) (wheat starch waste)**	***ULP (g/km)***
Upstream						
CO	218.3	218.0	11.2	301.2	13.0	*0.0904*
NO_x	-1.4	4.6	1.4	13.1	0.7	*0.4802*
NMVOC	2.1	1.8	1.7	4.6	1.6	*0.6692*
PM (Urban)	-5.2	86.8	97.7	97.3	94.7	*0.00706*
PM (Non-urban)	-5.6	-5.8	-15.3	1.5	-9.2	*0.00744*
Tailpipe						
CO	-26.9	-26.9	-26.9	-26.9	-26.9	*4.8500*
NO_x	5.0	5.0	5.0	5.0	5.0	*0.4614*
NMVOC	-14.4	-14.4	-14.4	-14.4	-14.4	*0.1678*
PM	-0.1	-0.1	-0.1	-0.1	-0.1	*0.00335*
Life cycle						
CO	**-22.4**	**-22.4**	**-26.2**	**-20.9**	**-26.1**	*4.9404*
NO_x	**1.7**	**4.8**	**3.2**	**9.1**	**2.8**	*0.9416*
NMVOC	**-1.2**	**-1.4**	**-1.6**	**0.8**	**-1.6**	*0.8370*
PM	**-4.4**	**31.9**	**32.3**	**39.1**	**33.6**	*0.01785*

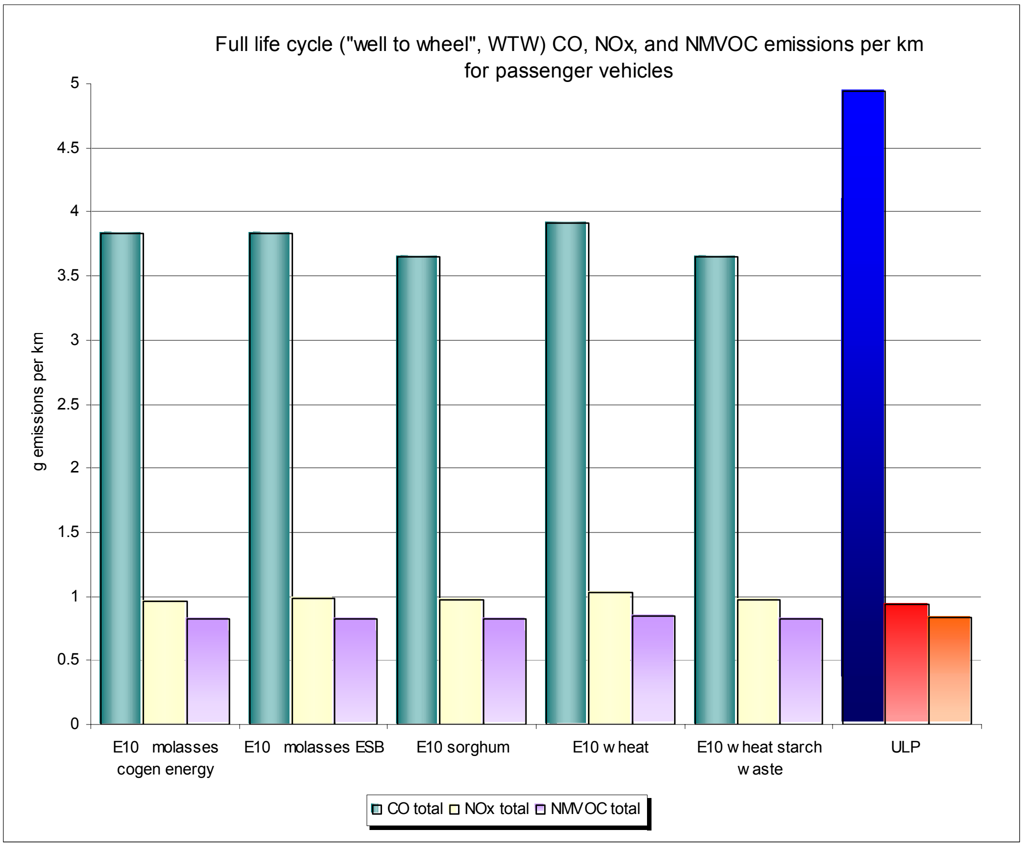

Figure 9. **Full life cycle CO, NO_x, and NMVOC emissions from E10 and ULP (passenger car)**

E10 Effects on PM

The use of ethanol as E10 in fuels is not expected to significantly alter the tailpipe emissions of PM10 (in the absence of relevant Australian data as discussed in Section 8.3.2.5), but the agricultural and industrial activities involved in the production and processing of the ethanol are expected to increase particulate emissions, on a life cycle basis. This is primarily a result of the particulate emissions involved in providing energy to the mill, the refinery and the distillery. If the energy is provided by cogeneration then the resulting particulate emissions are not ascribed to the ethanol and the particulate matter emissions are comparable, or even slightly lower, than those of petrol. In all other cases they are increased.

Figure 10 presents the PM upstream emissions per km, separate for emissions produced in the urban and non-urban areas, as well as the tailpipe emissions.

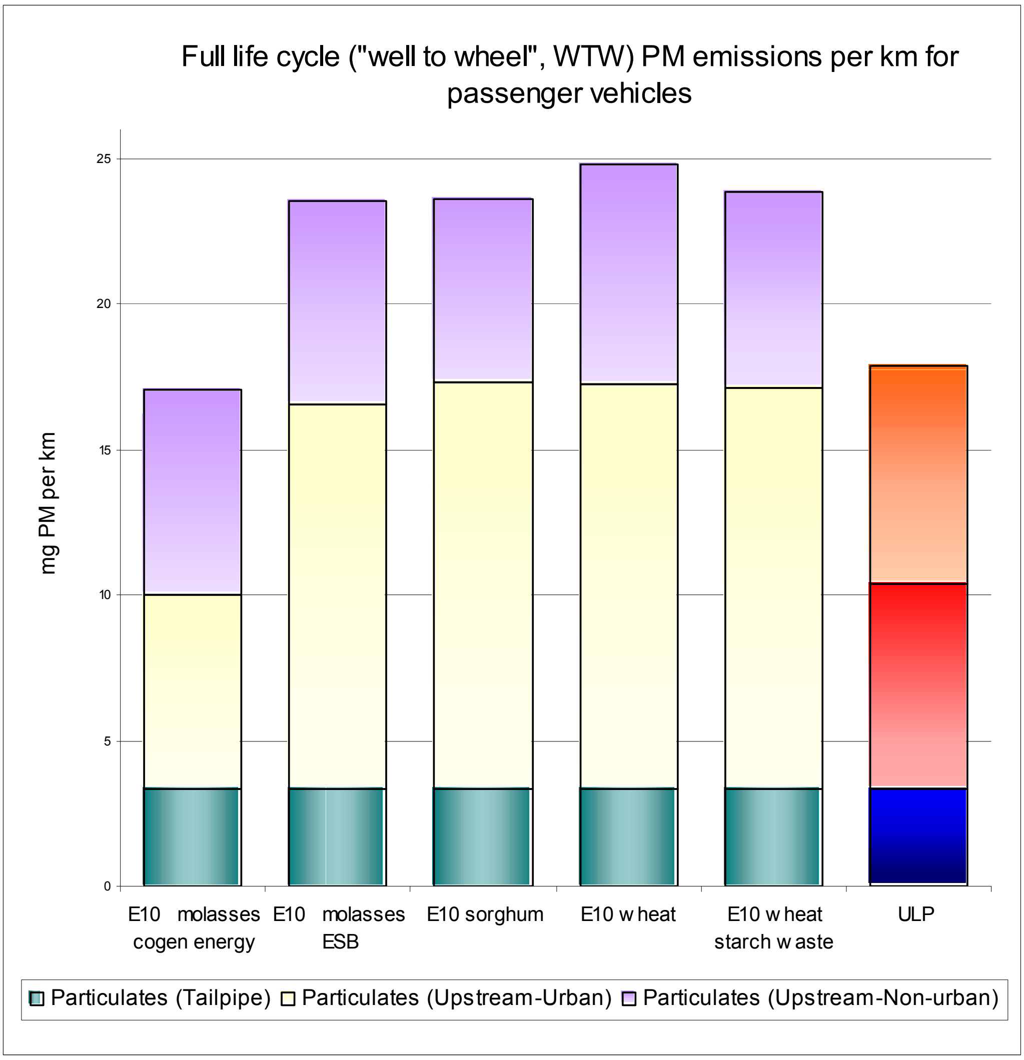

Figure 10. Full life cycle PM emissions from E10 and ULP separate for upstream-urban, upstream-non urban and tailpipe emissions (passenger car)

10.2 Environmental Performance of Biodiesel Compared to LSD, ULSD and XLSD

This Section presents the emission results per km for:

- the three types of diesel: low-sulfur diesel (LSD) (S < 500 ppm), ultra low-sulfur diesel (ULSD) (S < 50 ppm), and extra low-sulfur diesel (XLSD) (S < 10 ppm); and
- three types of biodiesel: canola oil, tallow, and waste oil.

The estimation of the environmental performance of biodiesel was based on combustion testing data presented by Graboski *et al.* (1999), Sharp (1998), and US EPA (2002). The variability in the emission rates described in the literature review is considered in the uncertainty analysis (Chapter 15). A normalisation process — to a typical vehicle — has then been carried out.

Average energy intensity rates used in the calculations have been adopted from Apelbaum (2002: Table 4.2.1-1, p.166): 12.63 MJ/km for buses, 12.24 MJ/km for rigid trucks, 5.11 MJ/km for 4WD, and 22.9 MJ/km for articulated trucks.

For upstream emissions of ULSD and XLSD the following assumptions were made:

- hydro-desulfurisation has been used to meet the 500 ppm sulfur limit (single-stage hydro-desulfurisation);
- further sulfur removal — down to < 50 ppm in the case of ultra-low sulfur diesel (ULSD) — requires a two-stage, hydro-desulfurisation using both the Co-Mo and Ni-Mo catalysts (hydro-processing 50% and hydro-cracking 50%); and
- extra-low sulfur diesel (XLSD) is manufactured using the hydro-cracking process (100%).

The results presented in this chapter are restricted to rigid trucks. A complete emission data set is, however, provided in Appendix VII, providing the greenhouse gases (GHG) and air pollutant emission rates for the other three categories of vehicles: 4WD, buses, and articulated trucks.

10.2.1 Greenhouse Gas Emissions per km (Rigid Trucks)

Table 31 presents the change (%) in the life cycle GHG emissions per km (for rigid trucks) resulted from using pure biodiesel compared to LSD, ULSD, and XLSD diesel base fuels. Table 32 shows the change (%) in GHG emissions for BD20 compared to LSD, ULSD, and XLSD diesel base fuels. Some comparative results are presented in Table 33. Tables examining the GHG emissions (per km) from 100% biodiesel as well as from each of the three diesel base fuels blended with 20% and 5% of the three biodiesel fuels, are presented in Appendix VII.

Table 31. Percentage change of full life cycle GHG emissions (CO_2-e) of BD100 relative to LSD, ULSD, and XLSD (rigid truck) (%)

GHG as CO_2-e (% change to each diesel type)	**Biodiesel (canola) BD100**	**Biodiesel (tallow) BD100**	**Biodiesel (waste oil) BD100**	***LS diesel***	***ULSD diesel***	***XLSD diesel***
To LSD						
GHG (Upstream)	341.7	306.8	-44.3	*base*	*10.77*	*20.69*
GHG (Tailpipe)	-98.9	-98.9	-98.9	*base*	*-2.18*	*-3.75*
GHG (life cycle)	**-22.96**	**-29.0**	**-89.5**	*base*	*0.05*	*0.46*
To ULSD						
GHG (Upstream)	298.8	267.2	-49.7	*-9.7*	*base*	*8.95*
GHG (Tailpipe)	-98.9	-98.9	-98.9	*2.2*	*base*	*-1.6*
GHG (life cycle)	**-23.0**	**-29.0**	**-89.5**	*-0.05*	*base*	*0.41*
To XLSD						
GHG (Upstream)	266.0	237.0	-53.9	*-17.14*	*-8.22*	*base*
GHG (Tailpipe)	-98.8	-98.8	-98.8	*3.9*	*1.63*	*base*
GHG (life cycle)	**-23.32**	**-29.3**	**-89.5**	*-0.46*	*-0.41*	*base*

Table 32. Percentage change of full life cycle GHG emissions (CO_2-e) of BD20 relative to LSD, ULSD, and XLSD (rigid truck) (%)

GHG as CO_2-e (% change to each diesel type)	**Biodiesel (canola) BD20**	**Biodiesel (tallow) BD20**	**Biodiesel (waste oil) BD20**
To LSD			
GHG (Upstream)	57.6	51.4	-10.42
GHG (Tailpipe)	-22.4	-22.4	-22.4
GHG (life cycle)	**-8.65**	**-9.7**	**-20.36**
To ULSD			
GHG (Upstream)	51.84	46.3	-9.5
GHG (Tailpipe)	-21.6	-21.6	-21.6
GHG (life cycle)	**-7.62**	**-8.7**	**-19.32**
To XLSD			
GHG (Upstream)	48	42.93	-8.22
GHG (Tailpipe)	-21	-21	-21
GHG (life cycle)	**-6.7**	**-7.76**	**-18.34**

Table 33. Percentage change of full life cycle GHG emissions (CO_2-e) of BD5 relative to LSD, ULSD, and XLSD (rigid truck) (%)

GHG as CO_2-e (% change to each diesel type)	**Biodiesel (canola) BD5**	**Biodiesel (tallow) BD5**	**Biodiesel (waste oil) BD5**
To LSD			
GHG (Upstream)	13.9	12.3	-3.08
GHG (Tailpipe)	-4.86	-4.86	-4.86
GHG (life cycle)	**-1.6**	**-1.9**	**-4.55**
To ULSD			
GHG (Upstream)	14.27	12.9	-1.05
GHG (Tailpipe)	-4.9	-4.9	-4.9
GHG (life cycle)	**-1.5**	**-1.5**	**-4.18**
To XLSD			
GHG (Upstream)	15.17	13.9	1.16
GHG (Tailpipe)	-4.1	-4.1	-4.1
GHG (life cycle)	**-0.1**	**-0.4**	**-3.04**

The main findings were:

- the upstream processes of growing and harvesting canola lead to high GHG emissions (4.4 times higher than LSD, 3.7 times higher than XLSD);
- the tailpipe GHG emissions are almost zero, which results in a saving in GHG emissions between 23% and 90% (or 230 g CO_2-e/km when replacing any type of diesel with BD100 canola, 289 g CO_2-e/km when replacing with BD100 tallow, and 894 g CO_2-e/km when using BD100 waste oil) – see Appendix VII;
- the extra upstream processing required for reducing the sulfur content results in higher GHG emissions for ULSD and XLSD;
- the highest savings in GHG emissions are obtained by replacing base diesel with biodiesel from waste oil (894 g CO_2-e/km for LSD to 898 g CO_2-e/km for XLSD);
- the large difference between the upstream emission of tallow and waste oil are based on the assumption that the tallow is being taken from existing market uses and is not a waste product, whereas the waste oil is taken to be a true waste, with no existing market. If low-grade tallow, with no other viable markets, was available, its emission profile would be the same as that of the waste oil;
- when blends with 20% biodiesel are analysed, the highest savings in GHG emissions are again obtained by replacing base diesel with biodiesel from waste oil: savings of 20.4% when the base fuel is LSD and 18.34%, when the base fuel is XLSD (or, in absolute values, 204 g CO_2-e/km for LSD base diesel to 180 g CO_2-e/km for XLSD base diesel);
- in case of 20% tallow blends, the life cycle GHG savings per km from the use biodiesel are in the range 9.7% to 7.7% (or 97 to 77 g/km) when the base fuel changes from LSD to XLSD;
- in case of 20% canola blends, the life cycle GHG savings per km vary between 8.65% to 6.7% (86 and 67 g/km) when the base fuel changes from LSD to XLSD; and
- blends with 5% biodiesel lead to much smaller GHG savings: the average savings are between 45 g CO_2-e/km to 30 g CO_2-e/km for waste oil blends when base diesel fuel changes from LSD to XLSD; 19 to 4 g CO_2-e/km for tallow biodiesel; 16 to 1.4 g CO_2-e/km for canola oil biodiesel.

Figure 11 presents the full life cycle (WTW) GHG emissions (g/km) from biodiesel BD100, BD20, and BD5, compared to LSD (rigid truck) partitioned into upstream and tailpipe emissions. The life cycle analysis results for GHG emissions, relative to ULSD and XLSD, are provided in Figure 12 and Figure 13.

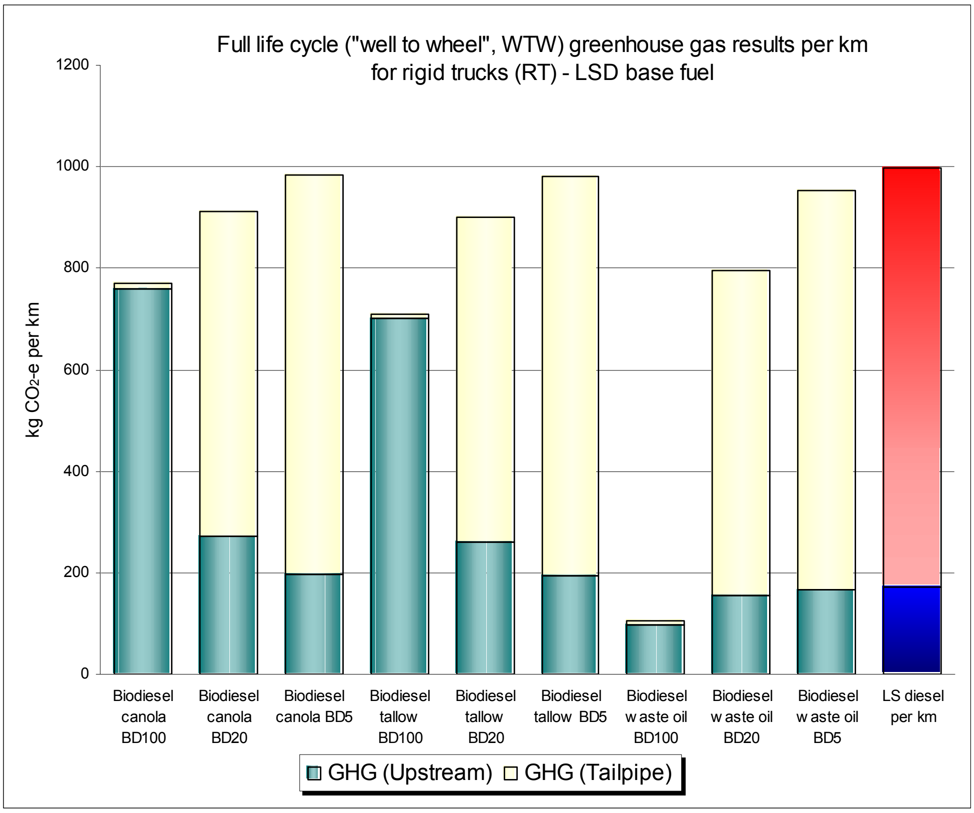

Figure 11. Full life cycle GHG emissions from biodiesel BD100, BD20, and BD5, compared to LSD (rigid truck)

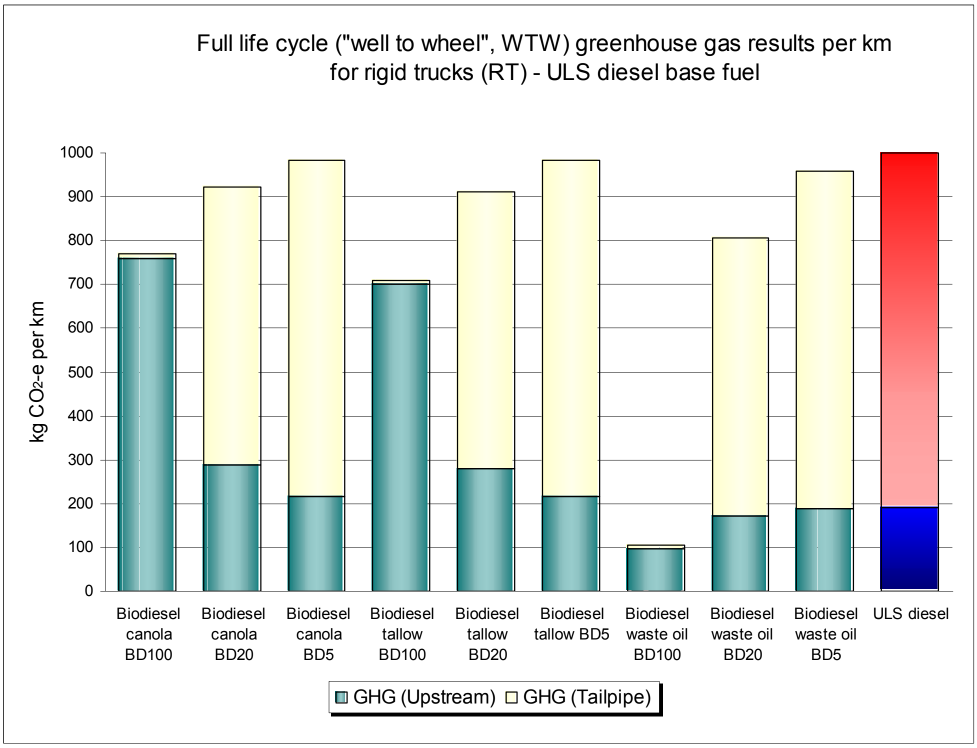

Figure 12. Full life cycle GHG emissions from biodiesel BD100, BD20, and BD5, compared to ULSD (rigid truck)

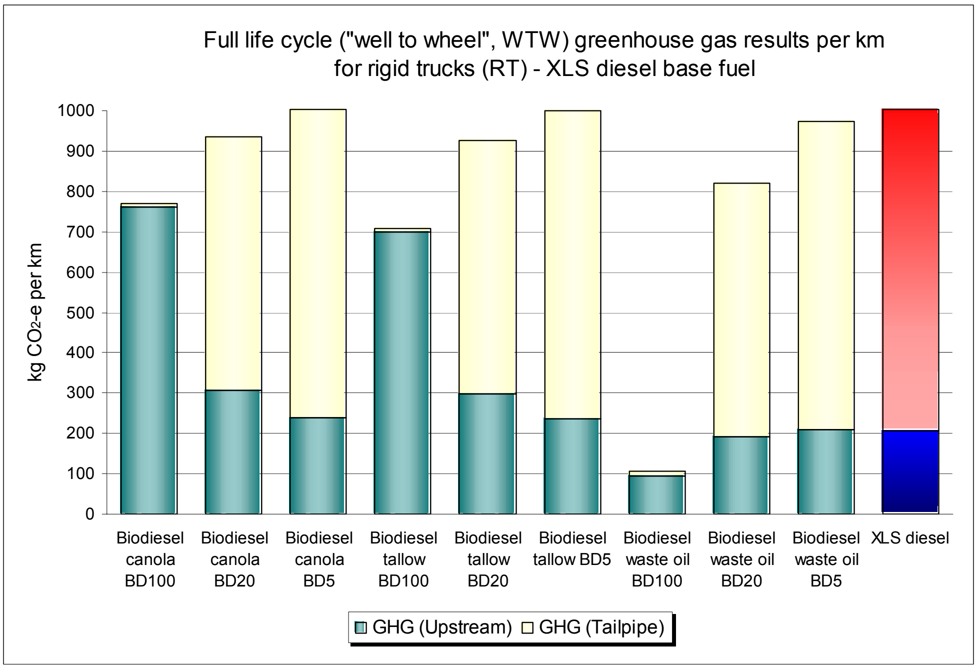

Figure 13. Full life cycle GHG emissions from biodiesel BD100, BD20, and BD5, compared to XLSD (rigid truck)

10.2.2 Air Pollutant Emissions per km (Rigid Trucks)

Table 34, Table 35 and Table 36 present the change (%) in the air pollutant emission rate (per km) of CO, NO_X, NMVOC, and PM for BD100, BD20 and BD5, respectively, in relation to LSD, ULSD, and XLSD diesel base fuels. Tables examining the emissions per km from 100% biodiesel as well as each of the three diesel base fuels blended with 20% and 5% of the three biodiesel fuels are presented in Appendix VII. In addition, process trees corresponding to canola, tallow, and waste oil biodiesel manufacture, transport and use are given in Appendix VII, presenting the WTW air pollution emissions per km for BD100.

Table 32 to Table 34 present the full life cycle air pollutant emissions on a per km basis for biodiesel canola, tallow, and waste oil (20% blend) compared to low sulfur diesel (LSD), ultra-low sulfur diesel (ULSD), and extra low-sulfur diesel (XLSD).

Table 34. Percentage change of full life cycle air pollutant emissions (as g/km) from BD100, LSD, ULSD, and XLSD (rigid truck) (%)

Impact category (full life cycle) (% change to each diesel type)	Biodiesel (canola) BD100	Biodiesel (tallow) BD100	Biodiesel (waste oil) BD100	LS diesel	ULSD diesel	XLSD diesel
To LSD						
CO	-27.43	-36.86	-47.04	base	-0.26	-1.6
NO_x	6.23	4.9	-5.35	base	-9.04	-18.25
NMVOC	-32.2	-35.04	-49.77	base	-8.26	-12.75
PM	-32.03	-32.6	-38.64	base	-19.91	-23.56
To ULSD						
CO	-27.24	-36.7	-46.91	0.26	base	-1.35
NO_x	16.79	15.33	4.1	9.94	base	-10.12
NMVOC	-26.11	-29.2	-45.24	9.0	base	-4.9
PM	-15.14	-15.83	-23.4	24.86	base	-4.55
To XLSD						
CO	-26.25	-35.83	-46.2	1.63	1.37	base
NO_x	29.94	28.31	15.8	22.32	11.26	base
NMVOC	-22.32	-25.55	-42.43	14.61	5.14	base
PM	-11.1	-11.82	-19.73	30.82	4.77	base

Table 35. Percentage change of full life cycle air pollutant emissions of BD20 relative to LSD, ULSD, and XLSD (rigid truck)

Impact category (full life cycle) (% change to each diesel type)	Biodiesel (canola) BD20	Biodiesel (tallow) BD20	Biodiesel (waste oil) BD20
To LSD			
CO	-17.0	-18.68	-20.47
NO_x	-4.71	-4.94	-6.74
NMVOC	-19.75	-20.25	-22.84
PM	-14.33	-14.42	-15.5
To ULSD			
CO	-16.08	-17.74	-19.54
NO_x	2.51	2.25	0.27
NMVOC	-13.18	-13.72	-16.54
PM	-4.37	-4.5	-5.81
To XLSD			
CO	-14.13	-15.81	-17.63
NO_x	12.53	12.24	10.04
NMVOC	-10.34	-10.91	-13.88
PM	-5.75	-5.87	-7.27

Table 36. Percentage change of full life cycle air pollutant emissions of BD5 relative to LSD, ULSD, and XLSD (rigid truck)

Impact category (full life cycle) (% change to each diesel type)	Biodiesel (canola) BD5	Biodiesel (tallow) BD5	Biodiesel (waste oil) BD5
To LSD			
CO	-14.35	-14.77	-15.21
NO_x	-3.96	-4.02	-4.47
NMVOC	-15.33	-15.45	-16.1
PM	-2.72	-2.77	-3.02
To ULSD			
CO	-13.41	-13.82	-14.27
NO_x	6.41	6.35	5.85
NMVOC	-8.17	-8.3	-9.01
PM	-2.14	-1.85	-2.17
To XLSD			
CO	-11.27	-11.69	-12.14
NO_x	10.9	10.81	10.27
NMVOC	-4.92	-5.07	-5.8
PM	0.08	0.06	-0.28

The main findings for the pure biodiesels were that:

- CO, NMVOC, and PM emissions from pure biodiesel are lower than those from all the diesel base fuels;
- NO_x emissions from pure canola and tallow biodiesel were higher than for all diesel fuels, and the difference increased with the reduction in sulfur content (less than 6% or 1g/km for canola, but more than 30% or 3 g/km for XLSD)
- NO_x emissions from waste oil biodiesel were lower than from LSD with 5%, but higher than ULSD with 4% or than XLSD with 16%;
- PM emissions from canola and tallow biodiesel are 32% lower than the emissions from LSD, 16% than the emissions from ULSD, and 11-12% lower than the emissions from XLSD; and
- the range of reductions in PM emissions from use of waste oil is 39% (LSD) to 20% (XLSD).

Similar findings were obtained with 20% canola biodiesel:

- CO, NMVOC, and PM were reduced when replacing diesel (regardless of the sulfur content) with 20% canola BD20; these benefits increase from canola oil to waste oil biodiesel, but decrease when the sulfur content decreases (the benefits from LSD to XLSD base diesel fuels);
- on average, the CO emissions benefit diminished from about 0.6 g/km for LSD to 0.5 g/km for XLSD for all biodiesel fuels;
- NMVOC reductions were 0.3-0.35 g/km for biodiesel blends compared to LSD and 0.1-0.2 g/km when the base diesel fuel was XLSD;
- PM emission benefits diminished considerably when the base diesel fuel had sulfur < 50 ppm (the 65-70 g/km PM benefits using BD20, relative to LSD, were reduced to 20-25 g/km relative to ULSD and XLSD); and
- BD20 NO_x emissions were 4% lower than from LSD, but higher than the ULSD (2%) and XLSD emissions (10-12%).

For 5% biodiesel blends, CO and NMVOC emission benefits were still evident, but NO_x emissions were higher for biodiesel blends than for ULSD or XLSD, and PM emissions are similar to those from diesel. This suggests that 5% blends may not be competitive with diesel beyond 2005, because after 1 January 2006 the standards for sulfur content become less than 50 ppm. After 2010 the standards for sulfur content are expected to be less than 10 ppm, in which case the biodiesel blends become even less preferential.

The results presented above suggest that biodiesel obtained from waste oil is a more environmentally friendly fuel than canola and tallow feedstocks (for combustion in rigid trucks).

General conclusions from this analysis were:

- all criteria air pollutants except NO_x were significantly decreased when replacing LSD with biodiesel;
- CO and NMVOC emissions were lower for all types of biodiesel – pure or blend – when compared to ULSD, but NO_x emissions from biodiesel were higher;
- with diesel sulfur contents less than 50 ppm, only pure biodiesel or 20% biodiesel blend had lower PM emissions than diesel; and
- the 5% biodiesel blend was less environmentally friendly than ULSD in terms of particulate matter.

The further reduction in sulfur content accentuated the increase in NO_x emissions between XLSD and biodiesel and diminished the benefits of CO and NMVOC.

10.3 Air Toxics

It is difficult to accurately assess the impacts of biofuels on air toxics emissions, due to the limited data available and the restricted range of air toxic compounds analysed in the studies.

10.3.1 E10 Effects on Air Toxics

The air quality in urban atmospheres is dependent on: primary pollutant emissions, formation of secondary pollutants, meteorological conditions and topography factors. A number of compounds emitted from vehicles negatively impact on air quality in urban environments. Emissions of NO_x and HC are involved in photochemical smog formation, some HC species can cause an odour problem and some are confirmed or suspected carcinogens. Compounds in smog contribute to respiratory and eye irritations and particles reduce visibility. The impact of E10 on the secondary production of ozone from compounds emitted by vehicles is discussed in the following section.

The comparative risks or benefits to air quality from the use of E10, relative to petrol, are difficult to assess. Combustion of E10 results in: increased tailpipe emissions of aldehydes (particularly acetaldehyde), although the base emissions are from low relative levels, and decreases in aromatic compounds (Section 8.3.2.4). Acetaldehyde and formaldehyde are products of the oxidation of HC (including ethanol). Acetaldehyde is the major oxidation product of atmospheric reactions of ethanol, with smaller amounts of formaldehyde (and glycolaldehyde) produced. Both aldehydes have very significant roles in atmospheric chemistry due to their role in formation of nitric acid and, PAN and other smog components (de Andrade *et al.*, 1998). The photochemical oxidation of ethanol and the reaction with ozone and NO_3 radicals are slow and of negligible importance. The products of ethanol reaction in the presence of NO_x are acetaldehyde and PAN (Kirchner *et al.*, 1997; de Andrade *et al.*, 1998). The California Environmental Policy Council (in conjunction with CARB) reviewed the air quality impacts from increased use of ethanol as an oxygenate, related to the MTBE phase-out (CARB, 1999). The report concluded that, compared with other non-MTBE gasoline components, the atmospheric formation of toxic compounds from ethanol is slow. The report stated: "*our findings from theoretical calculations using air-shed models with state-of-the-science chemistry* *indicate that other components of gasoline, such as aromatic compounds and olefins, are primarily responsible for the formation of formaldehyde, acetaldehyde, and PAN due to both their greater abundance in gasoline and their shorter atmospheric lifetimes."*

Data from air quality studies in Brazilian urban environments, where ethanol is commonly used as a fuel additive (neat ethanol or 20-25% ethanol blends dominate), have shown very high acetaldehyde levels (with a smaller relative increase in formaldehyde) (Correa *et al.*, 2003).

The major health issues are, however, the acute and chronic toxicities of the fuel blends and their products of combustion with the major public concern of toxicity being the carcinogenic effect. It is unlikely that exposure to ethanol vapours or skin contact, using E10 fuel, would cause acute toxicological problems greater than that from conventional petrol. The only significant potential damaging effect could be the increased aldehyde levels in the exhaust gases. At the levels present in vehicle exhaust, it is difficult to determine what the toxicological effects or potential carcinogenic properties are likely to be of acetaldehyde and formaldehyde. The acute and chronic exposure effects of individual compounds, and potential synergistic relationships between compounds, are similarly difficult to assess. The impact from the multitude of variables associated with catalytic converters is also uncertain, as both aldehyde conversion and production are catalytically possible. Vehicles using E10, compared with petrol, produce lower levels of some toxic compounds (e.g., benzene and PAH), but increase of others (e.g. acetaldehyde) and, therefore, make the comparison of the effect of HC emissions very intricate to estimate.

At present, the health costs of air toxics such as acetaldehyde and benzene are incorporated into costings and valuations for hydrocarbon emissions as given in Chapter 12.

The South Australian government, as part of its Environment Protection (Motor Vehicle Fuel Quality) Policy of 2002 used an "Air Toxicity Index", based on the cancer potency factors determined by the Californian EPA's Office of Environmental Health Hazard Assessment. The Californian EPA has compiled the most comprehensive list of compound carcinogenicity. Applying this air toxicity index to the tailpipe emissions (as mg/km) from the Petrohol study, an air toxic index was calculated (for the compounds formaldehyde, acetaldehyde, benzene and 1,3-butadiene) as shown in Table 37. Based on this method, the air toxicity index is reduced for E10 relative to the base fuels LP and ULP by 19% and 17%, respectively. This analysis does not include the impact of evaporative emissions, or the air toxic implications of atmospheric reaction products of the emissions, on the air toxic index, but the impact of the increased evaporative emissions from E10 (benzene) would be expected to offset, to some extent, these calculated improvements in air quality.

Table 37. Impact of tailpipe toxic emissions from petrohol for Pre- and Post-1986 vehicles using Air Toxic Index (ATI) derived from air toxic unit risk factors (California EPA)

	Form-aldehyde	Acet-aldehyde	1,3-Butadiene	Benzene	Total ATI	% ATI Difference
Unit Risk Factor $(\mu g/m^3)^{-2}$	6×10^{-6}	2.7×10^{-6}	1.7×10^{-4}	2.9×10^{-5}		
Relative Potency	*0.035*	*0.016*	*1*	*0.170*		
Air Toxic Index						
PRE-1986 Petrol	1.115	0.121	18.600	11.021	30.857	
PRE-1986 Petrohol	1.413	0.385	14.020	9.061	24.879	19%
POST-1986 Petrol	0.197	0.036	1.350	2.441	4.024	
POST-1986 Petrohol	0.251	0.112	1.230	1.760	3.352	17%

In regard to air quality of Australian cities, CO is regarded as one of the least significant criteria pollutants as CO concentrations here are generally below current national ambient air quality standards.

Ethanol does not contain sulfur atoms and therefore blending ethanol with conventional petrol will decrease emissions of SO_2.

10.3.2 Biodiesel Effects on Air Toxics

Air toxic tailpipe emissions from biodiesel, in particular, could not be assessed due to the lack of sufficient data.

Sharp (1998) analysed transient exhaust emissions from three diesel engines: 1997 Cummins N14, 1997 DDC Series 50, and 1995 Cummins B5.9 on diesel, biodiesel, and 25% blend biodiesel with diesel. The results indicated that a similar C_1 to C_{12} mix of compounds was present in the exhaust when using neat biodiesel, BD20 or diesel, but the mass of the compounds was significantly reduced on biodiesel compared to diesel (50% in the N14 and B5.9 engines and 30% in the Series 50). The biodiesel and diesel exhaust hydrocarbons both had about the same reactivity in terms of ozone formation (5-6 g O_3 per g of hydrocarbons).

The tests on the Cummins B5.9 and Detroit Series 50 engines were run with and without oxidation catalyst. The catalyst had only a small effect on speciated hydrocarbon mass for B5.9 (20%) and none for the Series 50.

The 20% biodiesel blend demonstrated many of the trends of neat biodiesel, but proportionally smaller, according to the biodiesel concentration. More importantly, the biodiesel blend did not generate any compounds that were not already present with the neat fuels.

Similar findings have been presented by Graboski *et al.* from the Colorado Institute for Fuels and Engine Research in 1999. The tests were performed on 27 neat biodiesels (from seven feedstocks) and three BD20 blends using an 1991 DDC Series 60 on the EPA heavy-duty transient cycle.

Regarding air toxics, the aldehyde emissions from various biodiesels were not significantly different from the aldehyde emissions from the certification diesel fuel. There were also attempts to perform chemical analysis of gaseous hydrocarbons – both dilute and undiluted exhaust streams – but the qualitative findings suggested no difference between certification diesel and biodiesel.

Pan *et al.*[18]. (2000) analysed PAH and nitro-PAH (n-PAH) emissions from standard diesel (2D), a pure biodiesel (methyl ester or soybean oil), and BD20 from three diesel engines: 1997 Cummins N14, 1997 Detroit Series 50, and 1995 Cummins B5.9. The emissions were measured over the heavy-duty transient Federal Test Procedure (FTP). Total PAH and n-PAHs decreased with the concentration of biodiesel, more significantly when no catalyst was used. The biggest reductions appeared for Cummins N14 engine: 75% with pure biodiesel, 12% for BD20 (PAH) and 93% with pure biodiesel, 13% for BD20 in case of n-PAH. The emission levels on n-PAH compounds were significantly lower than PAH levels for all three engines (at least one order of magnitude).

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[18] The team, from Southwest Research Institute, includes Christopher Sharp. The analysis appears to detail the tests described by Sharp in 1998.

The most comprehensive study on the sensitivity of exhaust emissions from use of biodiesel blends is the US EPA (2002) study. This study included not only the regulated air pollutants from 39 studies[19], but also a small amount of data on gaseous toxics. The results were considered only *"preliminary and potentially indicative of the true effects"*, due to the limited nature of data (p.85). The study studied 21 mobile air toxics, including six metals, MTBE, and acetaldehyde, acrolein, benzene, 1,3-butadiene, ethylbenzene, formaldehyde, polycyclic organic matter, styrene, toluene, xylene.

The study found a significant correlation between the percentage change in aggregate toxics and biodiesel concentration in the blend: percentage change in total toxics = - 0.00158 x (% biodiesel).

In order to evaluate individual toxics, four approaches were adopted:
- correlation of mass ratio of toxic/HC with biodiesel concentration;
- correlation of percentage change in toxic emissions with biodiesel concentration;
- binomial analysis of increases/decreases in toxics for biodiesel blends; and
- difference in average toxics effects at 20% and 100% biodiesel.

The conclusions can be summarised as follows:
- the toxic/HC ratios for acetaldehyde, acrolein, benzene, formaldehyde, naphthalene, styrene, 1,3-butadiene, toluene increase with the percentage of biodiesel, while the ratios for ethylbenzene and xylene decrease;
- the percentage change in acetaldehyde, ethylbenzene, formaldehyde, naphthalene, xylene emissions diminish with the concentration of biodiesel; the coefficients vary between -0.001606 for acetaldehyde to -0.00697 for ethylbenzene;
- the effect of biodiesel on acrolein, n-hexane, and styrene cannot be quantified with confidence, but qualitatively styrene and acrolein may increase with biodiesel, and n-hexane is likely to decrease; and
- the analysis of the toxics effects at 20 and 100% biodiesel has shown inconsistent trends for benzene, 1,3-butadiene, and toluene.

From these studies we may conclude that total air toxics, PAH and n-PAH emissions decline with biodiesel. Aldehydes appear to diminish, or stay the same, as Graboski *et al.* found (1999), but a study cited by Deni Greene Consulting Services (2002) – Krahl (1997) showed a 20% rise in aldehydes (p.29). Inconsistent results also appear for benzene, 1,3-butadiene, and toluene (increase or decrease).

All these results show that more research is required to identify the potential effects of biodiesel on the air toxics, accounting in the same time for all factors such as: type of vehicle, driving cycle/test, type of biodiesel, and of biodiesel in the blend.

10.4 E10 Effects on Ozone Formation

Ozone is not emitted directly from vehicles but is produced in the atmosphere from reactions between NO_x, HC and sunlight.

The effects of E10 on ozone formation are ambiguous because ozone formation is related to the particular airshed, being dependent on: meteorological conditions; topography, and the emissions of ozone precursors (NO_x and VOC) from all sources (major sources are motor vehicles, industry, and biogenics). To reduce the complexity associated with ozone formation,

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[19] The Sharp (1998) and Graboski *et al.* (1999) studies were included in the US EPA analysis.

the California Air Resources Board determines ozone reactivity from vehicle exhaust using the Maximum Incremental Reactivity (MIR) method, which is governed by the mass and composition of the non-methane organic compounds (NMOC) emissions.

The CSIRO component of the Petrohol Study (APACE, 1998) conducted HC speciation of exhaust samples from 11 vehicles and estimated the effects of E10 on OFP (using the MIR scale) and found a decrease of approximately 20%, relative to conventional petrol (Table 38). This effect was attributed largely to lower mass emissions of NMOC from E10 because the "reactivities" (mg O_3/mg NMOC emitted) of the exhaust emissions were found not to vary significantly between E10 and conventional fuelled vehicles. Ethanol emissions were, however, not included in this analysis but would not be expected to significantly affect OFP as ethanol is readily oxidised and has a low relative MIR value (1.34).

Table 38. Tailpipe emissions (mg/km) of aldehydes and air toxics (and calculated OFP) from Pre- and Post-1986 vehicles with petrol and petrohol

	NMOC [a]	OFP [b]
PRE-1986 Petrol	1694	7341
PRE-1986 Petrohol	1352	5975
POST-1986 Petrol	339	1341
POST-1986 Petrohol	266	1037

[a] Non Methane Organic Compounds (NMOC) calculated from total of approximately 50 compounds (HC and oxygenated organics)
[b] Ozone Forming Potential (OFP) (mg O_3/km)

The effects of ethanol in petrol on ozone formation have been extensively studied in the USA, with respect to replacing the MTBE oxygenate. The relevance of these studies to Australia is difficult to assess but qualitative information can be utilised. A study compared the emissions using an E10 fuel (high RVP) to an MTBE containing fuel (normal RVP) and found that OFP increased by 17% (CARB, 1998), primarily as a result of a 40% increase in evaporative emissions with E10. Other overseas studies (NRC, 1999) are ambiguous as to whether the addition of 10% ethanol is beneficial or is detrimental to air quality in terms of ozone. Howard (in NSTC, 1997) concluded that the addition of ethanol would result in adverse ozone impacts associated with increased NO_x and VOC emissions. According to the American Coalition for Ethanol, many US studies have concluded that the overall OFP of ethanol fuel blends is the same or lower than that of petrol (American Coalition for Ethanol Web Site).

At the November 2003 Clean Air Conference in Newcastle, speakers from both the NSW EPA and Queensland EPA reported that modelling studies had shown slightly reduced smog episodes for Sydney and Brisbane when E10 is used in the vehicle fleet.

The effect of the evaporative emissions contribution to total OFP from E10 is discussed further in the following section.

10.4.1 E10 Evaporative Effects on Ozone Forming Potential (OFP)

Evaporative ethanol emissions were not measured in the NSW EPA component of the APACE study (APACE, 1998), and therefore the effect of E10 evaporative emissions on OFP cannot be fully assessed. This is further complicated by the effect of ethanol (could be present in significant concentrations in evaporative emissions) on the HC measurement methods used, which could result in the underestimation of the HC evaporative emissions (Section 8.3.3.2).

Other studies have conducted evaporative emissions tests using E10. A 1990 study (Warner-Selph and Harvey, 1990) conducted evaporative emission tests on 5 cars (wide range of emission control systems) using an E10 fuel and found that overall the evaporative emissions increased by approximately 30%. They also tested for the unregulated evaporative emissions of MTBE, ethanol and benzene (using the FTP) and found no statistically significant change in MTBE or benzene emissions but found an increased ethanol concentration by 1 to 2 orders of magnitude. Grisanti *et al.* (1995) also showed that the direct evaporation of E10 leads to emissions enriched in ethanol relative to the original fuel. It is reasonable to expect that evaporative emissions from vehicles would also have an enhanced ethanol concentration despite the difficulty in determining the impacts of the material source on vapour composition. This is supported by a CARB study (CARB, 1998), which showed a higher proportion of ethanol in the vapour (increased by 25 to 50 wt.%) of the hot soak evaporative emissions using an E10 blend (RVP 7.8 psi). Similar results were found in a 1980 study (Furey and King, 1980) and were attributed to the high permeability, to ethanol, of the rubber fuel hoses. The CARB reported overall increases of 40% in evaporative emissions in an E10 blend using multi-day test procedures and estimated that use of such a blend would result in an overall increase of about 17% in OFP, relative to a fully complying gasoline (RVP less than 7 psi or 48 kPa). On this basis they have recommended against the use of 10% ethanol blends. Similarly the NRC (1999) concluded that the use of an ethanol-containing fuel with a RVP 1 psi higher is likely to produce a negative air quality impact.

By contrast, the US EPA (US EPA, 2001) have proposed an adjustment to the reformulated gasoline VOC standard to encourage the use of ethanol blends given the beneficial impacts of ethanol on CO emissions, in particular. It should be noted, however, that this increased use is associated with strict controls on the volatility of the gasoline with which the ethanol is blended.

10.5 Other Environmental Impacts of Biofuels

Recent work on a broad sustainability framework for the assessment of sustainability in bioenergy development (Keating and Vella, 2003) has identified six criteria against which sustainability can be assessed. These are:

1. Greenhouse Gas Impacts
2. Air Quality Impacts
3. Economic Impacts
4. Land and Water Impacts
5. Biodiversity Impacts
6. Social and Community Impacts

This review has explored: (1) greenhouse, (2) air quality, and (3) economic impacts in some depth. The review has also addressed some aspects of (6) social and community impacts particularly with respect to regional economies and employment. In this section the criteria (4) and (5) are addressed, namely the likely impacts of the 350 ML biofuels target on land and water resources and on biodiversity conservation.

Only "additional" environmental impacts arising specifically out of biofuel production are considered. Impacts that would occur irrespective of any biofuel production are outside the scope of this review.

Key considerations in assessing "other" environmental impacts include:

- is the biofuel production likely to require additional land clearing and cultivation?
- is the biofuel production likely to require additional extraction of water resources from rivers or aquifers?

- are the production practices for feedstocks for biofuels likely to lead to additional degradation of soil and water resources through salinity, erosion, acidification, chemical residue accumulation, nutrient and pesticide loss, etc?
- will waste products from the biofuel production process be disposed of or recycled in environmentally sustainable ways?

10.5.1 Ethanol from Waste Starch

We anticipate no significant impacts of ethanol production from waste starch on land, water and biodiversity resources, provided the waste products of the ethanol distillery are carefully managed.

The starch to ethanol segment of the current biofuels industry uses "non-commercial" starch grades and is not a driving force for feedstock production in its own right. Hence there will be no additional land clearing, water extraction or soil degradation issue specifically attributable to this biofuel. In the absence of an ethanol distillery, low-grade waste starches pose a waste disposal challenge (generally addressed via irrigation of agricultural land). The ethanol plant results in starch-rich wastes being displaced by nutrient rich wastes and these can be used to produce irrigated forage and silage (as is the current practice employed by Manildra at Bomaderry), provided simple principles of balancing nutrient inputs and off-take are followed.

10.5.2 Ethanol from Sugarcane Molasses

We anticipate no significant additional impacts of ethanol production from sugarcane molasses on land, water or biodiversity resources. once again this conclusion is conditional on the assumption that the wastes from the distillery operations are carefully managed.

Utilisation of C molasses for ethanol would not in itself lead to additional land clearing, sugarcane cultivation, water extraction or soil degradation. There may be some minor secondary impacts arising from a restriction in the supply of C molasses as an animal feeding supplement, although the environmental implications of reduced molasses supply are uncertain but not likely to be large.

In the past, waste management has been the major environmental issue with the ethanol from molasses industry. The waste product of ethanol distilleries in the sugar industry (locally referred to as dunder) is rich in nutrients (particularly potassium) and does also generate a high biological oxygen demand in aquatic ecosystems. Dunder escaping to waterways has been associated with fish kills. Over the last 20 years a major effort has transformed dunder management. The dunder by-product of the Sarina ethanol distillery is now used very efficiently and effectively as a fertiliser supplement, being spread at low rates (often supplemented with other compounds to serve as a balance nutrient source) on the sugarcane lands surrounding the distillery.

10.5.3 Ethanol from Whole Grains

Development of an ethanol industry based on conversion of whole grains into ethanol (and perhaps other animal feed products) could potentially lead to an expansion or intensification of grain cropping in the region of the distillery. This expansion could be onto lands previously uncultivated and may be associated with significant negative impacts on biodiversity resources. Likewise there is the potential for water extraction from rivers or groundwater systems to be enhanced to irrigate grain crops as feedstocks to biofuel factories. This may have negative environmental consequences for rivers and landscapes. If the presence of a biofuel distillery made grain cropping more profitable, this may result in a shift to more intensive crop rotations and in some circumstances, lead to enhanced soil degradation. While these negative impacts are theoretically possible, there is little evidence to suggest they would happen in practice. Use of grain products for biofuels does not appear to be significantly more attractive than use of grain for human or animal consumption. Hence, new land development, irrigation use or increased intensity of cropping are unlikely to be driven by the presence of a biofuel industry, at least in the short to medium term.

The waste management principles outlined above for ethanol from waste starch or C molasses also apply here. With careful management using established technologies, the nutrient rich wastes from ethanol plants can be recycled onto agricultural lands without concerns of undesirable environmental impacts.

10.5.4 Biodiesel from Waste Products

Production of biodiesel from waste oils is unlikely to generate any negative impacts on land, water or biodiversity resources. If anything, there may be some environmental benefits if it means there is a reduction in waste oils being disposed of directly into the environment. As with all other forms of biofuel, any waste streams of the biofuel production process would need to be sustainably managed.

11 TRANSPORT FUEL USE AND EMISSIONS IMPACTS

11.1 Background

This chapter presents estimates of the transport impact of obtaining 350 ML of biofuels production and use in road transport by 2010. In estimating the transport impact, the analysis considers the change in biofuels use between a reference (or 'business-as-usual') fuel use case and a 350 ML biofuels consumption case.

The market viability analysis of domestically produced biofuels has been used to inform the reference case level of biofuels production and consumption, and also the likely mix of biofuels that would be produced in order to obtain 350 ML of biofuels use in road transport in 2010. The estimates of the emissions performance of biofuels in comparison with conventional fuels, presented in chapters 8, 9 and 10, were used in the BTRE vehicle use and emissions models to calculate the impact on total emissions of additional biofuel supply and use.

The terms of reference require that the study pay particular attention to the current and proposed fuel standards. The fuel use and emissions projections reported in this study take into account the likely impact of current and future fuel quality and vehicle emissions standards on the level and mix of future fuel consumption. In particular, the planned introduction of Euro III (petrol) and Euro IV (diesel) vehicle emissions standards will require increased production of low sulfur fuels. The major policy initiatives affecting fuel use were outlined in Chapter 6 and details of current and future fuel quality and vehicle emissions standards are listed in Appendix IV. The reference case fuel use and road transport emissions projections take into account already announced future changes in vehicle emissions and fuel quality standards and the Government's foreshadowed changes to the excise and assistance arrangements for transport fuels.

The environmental and health impact of achieving 350 ML of biofuel use in road transport in 2010 will depend on the mix of biofuels (ethanol and biodiesel) entering the fuel stream and where the fuels are consumed (i.e. where the pollutant emissions occur). Based on the relative production costs of biofuels from existing and new capacity, and for different feedstocks, it has been assumed that in the reference case total biofuels consumption would entail approximately 85 ML supply of ethanol from existing capacity and 30 ML of biodiesel produced from waste oil sources. Based on the market viability analysis, Chapter 7, it is assumed that in order to achieve 350 ML of biofuels consumption in road transport, there would be an additional 60 ML of ethanol derived from C molasses feedstock, 145 ML of ethanol from dedicated cereal grain feedstocks (sorghum and wheat) and 30 ML of biodiesel produced from waste oil.

An issue for the reference case is the extent to which ethanol may be used to meet the future demand for higher octane rated fuel. As already mentioned, ethanol can be used as a fuel extender and octane enhancer. More stringent future vehicle emissions standards and fuel quality standards are likely to increase future demand for higher-octane fuels. For refiners, meeting the demand for higher-octane fuel may require additional capital investment or the use of octane enhancing additives, one of which is ethanol. (Some domestic refiners would already be able to meet the increased demand for 95 RON fuel without additional investment or additives). However, for ethanol to be used widely as an octane enhancer there are some key requirements that would need to be satisfied, among them: cost and supply, reliability, environmental performance and marketability (consumer acceptance). Appendix III provides a further discussion of these issues. Because these issues are still to be resolved, it is too early to say definitively whether ethanol would be used widely as part of a general octane enhancement strategy or not. Consequently, for the transport analysis presented here it has been assumed that any additional ethanol use in transport fuels is purely as a fuel extender, and not for the purposes of meeting the future demand for higher octane rated fuels.

Where extra biofuels consumption occurs will influence the environmental impact from additional supply; with the environmental and health impacts greatest in urban areas, where population densities are greatest. It has been assumed in this report that most of the biofuels consumption would occur in major urban areas.

The transport analysis considers the full fuel-cycle emissions, i.e. emissions arising from activities necessary to bring the fuel to point of final use and the emissions arising from the consumption of fuel within the vehicle. We adopt the terminology used by Beer *et al.* (2001), referring to emissions arising from the extraction, production, transport, processing and distribution of fuels as “upstream” emissions, and emissions arising from consumption of the fuel in the vehicle, including evaporative losses, as “tailpipe” emissions. Total upstream and tailpipe emissions are also referred to as ‘exbodied’ emissions.

The structure of this chapter is as follows. Section 2 outlines the reference case fuel use and vehicle emissions projections. Section 2 provides estimates of projected fuel use and vehicles emissions if 350 ML of biofuels were obtained in 2010. The last section provides estimates of the change in emissions between the reference case and the 350 ML biofuels consumption case.

11.2 Scenario 1: ‘Reference Case’ Fuel Consumption and Vehicle Emissions

Based on the results of the viability analysis (Chapter 7), the reference case assumes 85 ML of ethanol (80 ML sourced from waste starch and 5 ML from C molasses) and 30 ML of biodiesel use in the road transport sector in 2010. Total projected road transport emissions for the reference case, are based on the reference case level of biofuel consumption and the BTRE (2003) projections of conventional fuel use, with adjustments to account for the energy embodied in the biofuel use. The BTRE’s (2003) road transport projections were prepared prior to the Australian Government’s 2003–04 Budget announcement (Australia. Treasury, 2003) of the staged reduction in production assistance to domestic ethanol producers from 1 July 2008 and did not consider the viability of biodiesel from waste oil. The BTRE (2003) projections assumed that ethanol consumption in road transport would increase from approximately 45 ML in 2000–01[20] to 67 ML in 2009–10, and continue to grow by about 5% per annum, to total 110 ML in 2019–20. For biodiesel, the BTRE (2003) projections implied almost nil use of biodiesel in road transport over the period to 2019–2020. For the reference case emissions projections, the BTRE (2002) petrol and diesel use projections have been adjusted to account for the reference case level of ethanol and biodiesel use so that the total amount of energy use remains unchanged.

The BTRE (2003) vehicle emissions projections were derived using fleet-based models of vehicle use and fuel consumption. The fleet based models allow for the impact of future improvements in new vehicle fuel efficiency and lower emission rates. The principal features of these models are described in Box 1. Appendix IX provides the BTRE (2003) base case projections of total fuel use, vehicle travel and total vehicle end-use (tailpipe) emissions.

For both the reference case and the case where there is 350 ML of biofuels use in road transport, it is assumed that the size of the fleet, the number of new vehicles, the rate of scrappage and vintage specific fuel efficiency of the fleet are the same as for the BTRE (2003) base case. In

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[20] Total current (2003) domestic ethanol production is approximately 75 ML per annum, of which approximately 50 ML is used in transport activity.

other words, it is only the mix of fuels consumed in road transport that varies between the reference case and the 350 ML biofuel use case.

In the BTRE (2003) base case, the total vehicle stock is projected to increase by 1.8% per annum, from 12.5 million vehicles in 2000 to 14.9 million vehicles in 2010. The stock of light vehicles (passenger cars, LCVs and motor cycles) is projected to grow by 1.8% per annum. The stock of heavy vehicles (rigid and other trucks, articulated trucks and buses) is projected to grow by 1.0% per annum.

The fuel efficiency of new passenger cars is projected to improve by 20% between 2000-2001 and 2019-2020. The on-road fleet average passenger car fuel efficiency is projected to improve by 5% over the same period. New commercial vehicle fuel efficiencies are assumed to remain unchanged over the projection horizon, reflecting improvements in engine technology but offset by increases in the average size of new commercial vehicles. Consequently, the on-road fleet average fuel efficiency of rigid trucks is projected to decline by 6% between 2000-2001 and 2019-2020.

Box 1: BTRE Fleet-Based Vehicle Emissions Models

The BTRE has developed fleet-based models of total road transport vehicle activity in order to incorporate the dynamic effects of changes in future vehicle fuel efficiency and emissions performance on total future emissions. The BTRE's vehicle fleet models incorporate vintage and age specific characteristics of the vehicle fleet, calculating vehicle utilisation and fuel consumption for each vintage over the life of the vintage. The models incorporate the effect of vehicle aging on the emissions performance of each vehicle vintage and allow for improvements in fleet average emissions performance through the scrappage of existing vehicles and entry of new vehicles each year. The models can be used to assess the impact on emissions of future changes in vehicle emissions and fuel quality standards.

The principal factors affecting the level of vehicle emissions essentially are the size of the vehicle stock, average vehicle utilisation, fuel efficiency and emissions characteristics of new vehicles entering the fleet, the rate of turnover of the fleet and the type of fuel used. The BTRE (2003) base case projections of the total vehicle stock, by vehicle type, and the fleet average on-road fuel intensity of different vehicle types are provided in Appendix IX. Further details about the assumptions used in the BTRE base case emissions projections are outlined in BTRE (2002, 2003).

The BTRE vehicle emissions models include emission rates for vehicular emissions of greenhouse gases and pollutants. Emission rates for current vehicle vintages are based on historical emissions test results, generally measured in terms of grams per vehicle kilometre travelled. For new vehicles, the emission rates are based on current model emissions test results and future vehicle emissions standards. The BTRE's vehicle emissions models allow for some deterioration in emissions performance of each vehicle vintage as the vehicles age, as well as higher 'on-road' emissions compared to emission test results. (Appendix IX outlines the vehicle emissions deterioration assumptions included in the BTRE passenger car and rigid truck vehicle emissions models.)

11.2.1 Emission Rates

The emissions rates for ethanol and biodiesel blend fuels used to compute the total vehicular emissions are based on the relative emissions rates, for petrol and ethanol blends for the former, and automotive diesel (LSD, ULSD and XLSD) and biodiesel for the latter. The ethanol and biodiesel emission rates are based on available emissions test evidence scaled to recent on-road vehicle emissions test results. As outlined above, the BTRE vehicle emissions models allow for planned improvements in new vehicle fuel consumption, vehicle emission performance and fuel quality standards over the projection horizon. In order to incorporate both the impact of likely future improvements in vehicle emissions and the impact on vehicle emissions due to substitution of biofuels for conventional fuels, the emission rates for ethanol blend fuels and biodiesel used in the BTRE vehicle models were set such that the relativity between emissions rates for conventional fuels and biofuels were preserved. The effect of this assumption is that the absolute change in emissions, attributable to additional biofuel use, generally declines over the projection horizon, in line with projected reductions in future fleet average emission rates.

11.2.2 Road Transport Sector Biofuel Use Assumptions

In order to simplify the transport impact analysis, it is assumed that all additional road transport ethanol consumption would be used as a 10% ethanol-90% ULP blend (E10) in passenger cars and all road transport sector biodiesel consumption would be consumed as pure biodiesel (BD100) in rigid trucks.

Passenger cars constitute almost 80% of the vehicle fleet and account for 77% of the total VKT. Passenger cars are also predominantly petrol fuelled, and so are likely to consume the majority of any ethanol sold as blended fuel. Likewise, rigid trucks are predominantly diesel fuelled undertaking the majority of urban freight operations and, along with urban buses, are likely to be the main early adopters of biodiesel.

As a further simplification, it has been assumed that all biodiesel use by rigid trucks would be consumed in urban areas. This assumption is based on the pattern of up-take observed for alternative fuels, such as LPG and CNG, in heavy-duty commercial vehicles. It has been the experience that alternative fuels are more likely to be used initially in particular urban applications where there are large fleets of vehicles with their own dedicated refuelling facilities, for example urban bus operations and waste management truck fleets. For alternative fuels, such as LPG and CNG, the capital cost of conversion and proximity to fuel supply are factors in the pattern of uptake. In those cases, the additional capital cost of modifying the vehicle to use the alternative fuels may lead to the take up by only a small number of users. Users of biodiesel, which require almost no special vehicle modification, will face no such costs and it may end up being used more widely. However, due to issues of consumer uncertainty, it may also be the case that biodiesel will only gain wider acceptance after some experience of use in particular applications. Also, constraints in the fuel distribution chain (e.g. limits on the number of fuel types that can be handled at service stations) would limit the availability of biodiesel. It is not expected that extending the use of biodiesel in buses would significantly alter the quantitative results.

Analysis of the impact of these assumptions on total emissions is outlined in Chapter 13. In particular, the sensitivity of the transport emissions impact of the assumptions about the biodiesel blend (BD100, BD20 or BD5) was tested.

11.2.3 Emissions Rates – Ethanol and Biodiesel

The upstream and tailpipe emission rates for passenger cars and rigid trucks, used in estimating the transport impacts, are listed in Table 39 and Table 40. Table 39 provides upstream and tailpipe greenhouse gas emission rates for ULP use by passenger cars, as well as the percentage change in emissions due to substitution of E10 for ULP. Table 40 provides upstream and tailpipe greenhouse gas emission rates for BD100, LSD, ULSD and XLSD use in rigid trucks, and the percentage change in emissions resulting from the substitution of waste oil for LSD, ULSD and XLSD.

Substitution of E10 for ULP in passenger cars results in less CO_2, CO, NMVOCs and PM and slightly more NO_x and CH_4 output. Without treatment to reduce the volatility of the base petrol stock, E10 is more volatile than either ULP or E100, resulting in higher evaporative emissions of VOCs. Use of BD100 in place of automotive diesel results in less greenhouse gas emissions and less output of CO, NMVOCs and PM, but higher NO_x levels. Relative to LSD, BD100 results in one-third less particle emission. However, relative to ULSD and XLSD, the reduction in PM emissions from BD100 is much less, 16.4% and 12% respectively.

Table 39. Passenger car emission rates – E10 and ULP

	Emission rates (g/km)						**Emissions performance relative to ULP (percent)**				
Impact category	E10 (ULP) (molasses cogen energy ESB)	E10 (ULP) (molasses ESB)	E10 (ULP) (sorghum ESB)	E10 (ULP) (wheat ESB)	E10 (ULP) (wheat starch waste ESB)	ULP	E10 (ULP) (molasses cogen energy ESB)	E10 (ULP) (molasses ESB)	E10 (ULP) (sorghum ESB)	E10 (ULP) (wheat ESB)	E10 (ULP) (wheat starch waste ESB)
Upstream emissions											
CO_2	58.84	64.65	69.62	70.58	65.75	56.21	4.7	15.0	23.9	25.6	17.0
CH_4	0.4933	0.5001	0.5213	0.5203	0.5087	0.5431	-9.2	-7.9	-4.0	-4.2	-6.3
N_2O	0.0047	0.0048	-0.0035	0.0104	0.00048	0.00033	1329.8	1359.0	-1152.3	3073.3	46.2
CO	0.2876	0.2873	0.1005	0.3625	0.1021	0.09036	218.3	218.0	11.2	301.2	13.0
NO_x	0.4733	0.5021	0.4867	0.543	0.4834	0.4802	-1.4	4.6	1.4	13.1	0.7
NMVOC	0.6833	0.6814	0.6803	0.6997	0.6799	0.6692	2.1	1.8	1.7	4.6	1.6
PM (Urban)	0.00669	0.01319	0.01396	0.01393	0.01375	0.00706	-5.2	86.8	97.7	97.3	94.7
Tailpipe emissions											
CO_2	316.5	316.5	316.5	316.5	316.5	340.5	-7.0	-7.0	-7.0	-7.0	-7.0
CH_4	0.00742	0.00742	0.00742	0.00742	0.00742	0.00716	3.7	3.7	3.7	3.7	3.7
N_2O	0.00227	0.00227	0.00227	0.00227	0.00227	0.00227	-0.1	-0.1	-0.1	-0.1	-0.1
CO	3.547	3.547	3.547	3.547	3.547	4.85	-26.9	-26.9	-26.9	-26.9	-26.9
NO_x	0.4846	0.4846	0.4846	0.4846	0.4846	0.4614	5.0	5.0	5.0	5.0	5.0
NMVOC	0.1437	0.1437	0.1437	0.1437	0.1437	0.1678	-14.4	-14.4	-14.4	-14.4	-14.4
PM	0.00334	0.00334	0.00334	0.00334	0.00334	0.003346	-0.1	-0.1	-0.1	-0.1	-0.1

Table 40. Rigid truck emission rates – biodiesel and automotive diesel

	Biodiesel (g/km)			Automotive diesel (g/km)			Performance relative to diesel (percent)		
Impact category	**Biodiesel (canola ESB)**	**Biodiesel (tallow ESB)**	**Biodiesel (waste oil)**	**LSD**	**ULSD**	**XLSD**	**Biodiesel (waste oil) : LSD**	**Biodiesel (waste oil) : ULSD**	**Biodiesel (waste oil) : XLSD**
Upstream emissions									
CO_2	451.30	410.00	92.27	145.6	164.2	181.4	-36.63	-43.81	-49.13
CH_4	1.084	1.002	0.162	1.249	1.243	1.234	-87.03	-86.57	-86.87
N_2O	0.923	0.868	0.00048	0.00085	0.001044	0.00123	-43.62	-54.06	-60.88
CO	0.89	0.548	0.1785	0.3234	0.352	0.3782	-44.8	-49.3	-52.8
NO_x	1.90	1.722	0.3485	1.165	1.303	1.431	-70.1	-73.25	-75.65
NMVOC	0.321	0.28	0.05508	0.4731	0.4806	0.4865	-88.36	-88.54	-88.68
PM (Urban)	0.017	0.016	0.00246	0.01725	0.01797	0.0186	-85.76	-86.33	-86.8
Tailpipe emissions									
CO_2	0.00	0.00	0.00	816.40	798.60	785.80	-100.0	-100.0	-100.0
CH_4	0.198	0.198	0.198	0.224	0.218	0.214	-11.61	-9.1	-7.65
N_2O	0.0163	0.0163	0.0163	0.01787	0.01754	0.0173	-8.78	-7.07	-5.62
CO	1.743	1.743	1.743	3.305	3.267	3.192	-47.26	-46.65	-45.4
NO_x	12.34	12.34	12.34	12.24	10.89	9.528	0.82	13.31	29.51
NMVOC	0.705	0.705	0.705	1.04	0.907	0.834	-32.2	-22.31	-15.44
PM	0.283	0.283	0.283	0.432	0.339	0.322	-34.48	-16.44	-11.97

11.2.4 Fuel Use Projections – Reference Case

In the reference case total production of fuel ethanol from waste starch is assumed to increase from around 50 ML today to 80 ML by 2010. The current production of around 2–5 ML of ethanol for use in transport from C molasses is assumed to continue in the reference case.

Current production plans would see at least 60 ML of biodiesel production capacity within the next few years[21], producing biodiesel primarily from waste oil feedstocks. Not all of the planned biodiesel production would necessarily be used in road transport. Some may be consumed in off-road transport uses or in other sectors of the economy, such as the mining sector, for example. Total production in 2003-04 is projected to be approximately 10–15 ML. The announced phase-out of the production assistance grants, offsetting excise on biodiesel, would likely affect planned production levels closer to the phase-out period. The reference case assumes an intermediate case: slight growth in current production, from approximately 10 ML in 2003–2004 to 30 ML in 2010, employing waste oil feedstocks.

Figure 14 illustrates the level of biofuels supply, by feedstock, assumed under the reference case. The figure also illustrates the assumed ethanol and biodiesel supply for the case where 350

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[21] In 2003 Biodiesel Industries Australia has commenced production from a plant at Rutherford (NSW) that has a capacity of 14-17 ML per year. The Australian Biodiesel Consultancy will commence production of biodiesel, based on a mix of waste oil and tallow feedstocks, in early 2004. The Australian Biodiesel Consultancy production facility will have the capacity to produce up to 40 ML of biodiesel per year.

ML of biofuel use in the road transport sector is attained in 2010. Note that the order of presentation does not imply any priority or ranking of cost effectiveness.

Figure 14. Projected biofuels mix: Reference case and 350 ML Biofuels consumption case

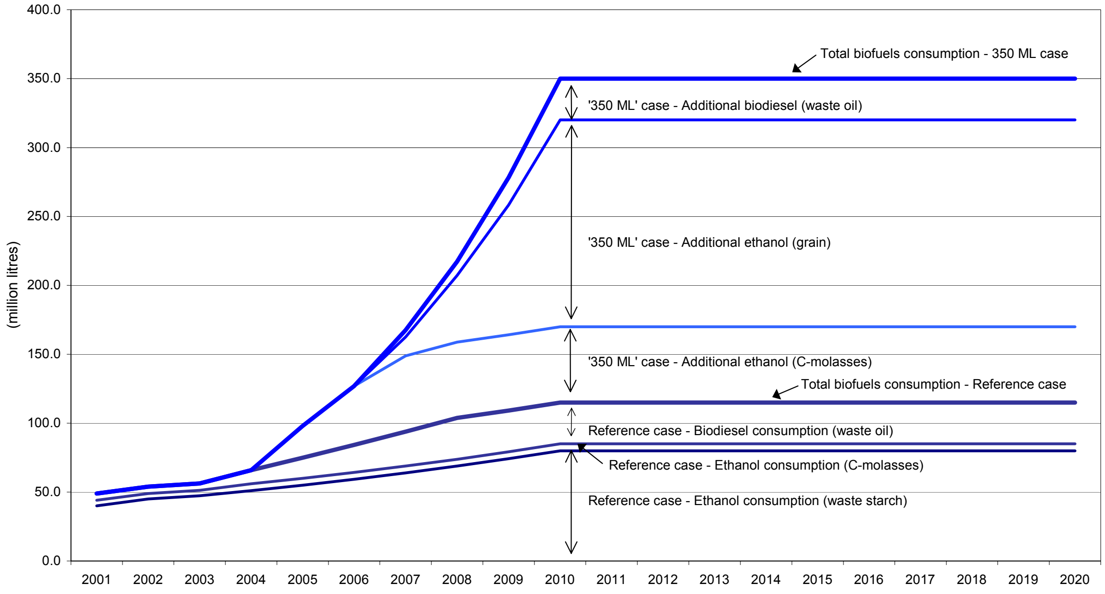

Table 41 shows the reference case fuel consumption projections for passenger motor vehicles from 2000–2001 to 2019–2020. The fuel use projections for all fuels other than biofuels are based on the BTRE (2003) projections. As already mentioned, the reference case ethanol consumption estimates differ slightly from those in BTRE (2003). In BTRE (2003) ethanol consumption by passenger motor vehicles was projected to increase from approximately 45 ML in 2002 to 67 ML in 2010, whereas in the current reference case ethanol consumption is projected to increase to 85 ML in 2010. For the current analysis, the reference case projections of ULP/PULP fuel consumption by passenger motor vehicles are based on the projections produced for BTRE (2003), adjusted for the energy equivalent amount of additional projected ethanol consumption in the reference case relative to BTRE (2003). The additional ethanol consumption assumed in the reference case has a negligible impact on the projected growth in total petrol consumption.

Table 41. Reference case passenger motor vehicle fuel consumption projections, by fuel type, 2000–2020 (million litres)

Year	LP / LRP	ULP / PULP	PULP	Total Petrol	ADO	LPG	NG[a] (Petrol equiv.)	Ethanol	Total
2000–01	2,932.3	11,875.2	774.6	15,582.0	650.0	1,450.0	9.0	40.0	17,731
2001–02	2,456.0	12,592.5	871.4	15,919.8	670.0	1,400.0	10.0	45.0	18,045
2002–03	2,081.4	13,406.0	980.3	16,467.7	680.0	1,450.0	11.9	50.0	18,660
2003–04	1,756.3	14,131.5	1,102.8	16,990.7	691.2	1,478.0	14.1	53.9	19,228
2004–05	1,464.5	14,766.2	1,240.7	17,471.4	702.7	1,506.6	16.7	58.2	19,756
2005–06	1,208.5	15,078.0	1,395.8	17,682.3	714.3	1,535.8	19.9	62.8	20,015
2006–07	1,018.3	15,307.2	1,570.3	17,895.8	726.1	1,565.4	23.6	67.7	20,279
2007–08	855.1	15,476.0	1,766.5	18,097.6	738.1	1,595.7	28.0	73.0	20,532
2008–09	727.2	15,565.7	1,987.4	18,280.2	750.3	1,626.6	33.2	78.8	20,769
2009–10	618.0	15,600.4	2,235.8	18,454.2	762.7	1,658.0	39.5	85.0	20,999
2010–11	472.0	15,599.4	2,515.3	18,586.7	775.3	1,690.1	46.9	85.0	21,184
2011–12	355.7	15,524.3	2,829.7	18,709.7	788.1	1,722.8	55.6	85.0	21,361
2012–13	263.8	15,363.6	3,183.4	18,810.8	801.1	1,756.1	66.0	85.0	21,519
2013–14	191.6	15,131.3	3,581.3	18,904.3	814.4	1,790.0	78.4	85.0	21,672
2014–15	135.4	14,818.2	4,028.9	18,982.5	827.9	1,824.7	93.1	85.0	21,813
2015–16	114.8	14,392.2	4,532.6	19,039.6	841.5	1,859.9	110.5	85.0	21,937
2016–17	97.6	13,905.6	5,099.1	19,102.4	855.5	1,895.9	131.2	85.0	22,070
2017–18	83.1	13,314.7	5,736.5	19,134.4	869.6	1,932.6	155.8	85.0	22,177
2018–19	70.9	12,627.2	6,453.6	19,151.8	884.0	1,969.9	185.0	85.0	22,276
2019–20	60.8	11,672.3	7,446.9	19,180.0	900.0	1,976.3	219.6	85.0	22,361

Notes LP/LRP = leaded petrol / lead replacement petrol (all sales in this category after 2001 are LRP).
ULP = unleaded petrol / premium unleaded petrol.
PULP = premium unleaded petrol / premium unleaded petrol.
ADO – automotive diesel oil.
LPG – liquefied petroleum gas.
NG – natural gas.

For simplicity, total motor vehicle sales of ethanol (as E10) have been allocated to car fuel use.

Source BTRE (2003b) and BTRE estimates.

Table 42 shows the projected fuel consumption by rigid trucks for the current study's reference case. Again, the rigid truck fuel consumption projections are based on the projections described in BTRE (2003). For biodiesel, BTRE (2003) assumed there would be no biodiesel use in the road transport sector in 2010. As in the case of ethanol, for the reference case the BTRE (2003) projected rigid truck diesel fuel consumption has been revised to account for the energy equivalent amount of diesel displaced by the assumed 30 ML of biodiesel consumption by the road transport sector in 2010.

Table 42. Reference case rigid truck fuel consumption projections, by fuel type, 2000–2020 (million litres)

Year	Petrol	ADO	LPG	NG[a] (petrol eq.)	Biodiesel (BD100)	Total
2000–01	76.1	1,754.2	60.0	0.4	5.0	1,895.8
2001–02	64.1	1,809.9	65.0	0.7	5.0	1,944.7
2002–03	54.6	1,824.3	70.0	1.0	5.0	1,954.9
2003–04	47.1	1,873.5	75.0	1.3	10.0	2,006.9
2004–05	41.2	1,879.9	75.8	1.4	15.0	2,013.2
2005–06	31.0	1,903.0	76.5	1.4	20.0	2,032.0
2006–07	26.5	1,926.2	77.3	1.5	25.0	2,056.5
2007–08	22.0	1,939.5	78.0	1.6	30.0	2,071.1
2008–09	19.5	1,954.2	78.8	1.7	30.0	2,084.2
2009–10	19.0	1,977.8	79.6	1.9	30.0	2,108.3
2010–11	18.5	1,993.4	80.4	2.0	30.0	2,124.3
2011–12	19.0	2,005.6	81.2	2.2	30.0	2,138.0
2012–13	19.5	2,016.7	82.0	2.4	30.0	2,150.6
2013–14	20.0	2,027.1	82.8	2.6	30.0	2,162.6
2014–15	20.5	2,035.7	83.7	3.4	30.0	2,173.3
2015–16	21.0	2,045.4	84.5	4.5	30.0	2,185.4
2016–17	21.5	2,064.1	85.4	5.8	30.0	2,206.8
2017–18	22.0	2,068.1	86.2	7.6	30.0	2,213.8
2018–19	22.5	2,068.2	87.1	9.8	30.0	2,217.6
2019–20	23.0	2,065.2	87.9	12.8	30.0	2,218.9

Notes: ADO – automotive diesel oil.
LPG – liquefied petroleum gas.
NG – natural gas.
For simplicity, total motor vehicle sales of ethanol (as E10) have been allocated to car fuel use.
Source BTRE (2003) and BTRE estimates.

11.2.5 Reference Case Vehicle Emissions

The reference case total end-use vehicle emissions for passenger cars and rigid trucks are as shown in Table 43 and Table 44. Greenhouse gas emissions from passenger cars are projected to increase 26% between 2000-01 and 2019-20. Most criteria pollutant emissions are projected to decline by between 18 (SO_x) and 43 (CO) % over the period 2000-2001 to 2019-2020. PM emissions from passenger cars are projected to increase slightly over the same period. Greenhouse emissions from rigid trucks are projected to increase by 17% between 2000-2001 and 2019-2020, and emissions of all pollutants are projected to decline between 29 (NO_x) and 48 (VOCs) % over the same period. (Due to the lower sulfur content, SO_x emissions are projected to fall by 93%.)

The estimates accord with the Kyoto Protocol convention that carbon dioxide emissions generated as a result of combustion of a fuel produced from a renewable source are not included in the calculation of total CO_2 emissions.

The BTRE vehicle emissions models adopt the IPCC method of accounting for carbon and assume that all of the carbon content of the fuel is counted either as CO_2 emissions or as solid products. This method of carbon accounting results in a slight double counting of the carbon content of CH_4, CO and NMVOC emissions. From a greenhouse perspective, this approach provides a satisfactory approximation as CH_4, CO and NMVOCs eventually convert to CO_2 in the atmosphere. To derive an estimate of actual CO_2 emissions released upon combustion, the carbon contained in the CH_4, CO and NMVOC emissions is subtracted from the total estimated CO_2 emissions (BTCE, 1995).

In computing passenger car and rigid truck vehicle emissions, it has been assumed that ethanol and biodiesel consumption is consumed in proportion with conventional fuel consumption by - each vehicle vintage. In other words, it is assumed that the extra biofuel supply is not consumed more heavily by either older or newer vehicles. This may affect the size of any increase or decrease in emissions from biofuels. This implicit assumption is necessary because the BTRE's vehicle emissions models do not currently allow the user to exogenously alter the mix of fuel use by age of vehicle. The impact of this assumption is that the absolute difference in vehicle emissions performance, arising from substitution of biofuels for conventional fuels, will generally decline over time, in proportion to changes in the fleet average vehicle emissions performance.

Table 43. Reference case passenger car vehicle emission projections, by emission type, 2001–2020 (kt)

	Greenhouse	**Air Pollutants**				
Year	**$CO_{2\text{-}e}$**[a]	**CO**	**NO_x**	**VOC**	**SO_x**	**PM**
2000–01	41,080	2,596.9	252.4	393.2	5,226.3	10.60
2001–02	41,929	2,508.3	249.9	380.9	4,977.7	10.74
2002–03	43,406	2,470.2	250.7	376.9	4,731.2	10.97
2003–04	44,794	2,429.6	250.5	375.8	4,562.4	11.23
2004–05	46,066	2,377.5	248.5	369.9	4,033.7	11.44
2005–06	46,672	2,298.3	242.6	359.9	3,906.1	11.49
2006–07	47,277	2,223.0	236.6	350.8	3,953.9	11.65
2007–08	47,858	2,150.2	230.4	346.1	3,999.1	11.80
2008–09	48,401	2,083.1	224.2	338.8	4,040.3	11.94
2009–10	48,924	2,015.5	218.0	332.2	4,079.7	12.05
2010–11	49,352	1,946.1	211.4	325.6	4,110.0	12.12
2011–12	49,759	1,874.8	204.3	319.8	4,138.5	12.15
2012–13	50,117	1,804.4	197.1	314.5	4,162.2	12.17
2013–14	50,465	1,737.0	190.4	310.0	4,184.3	12.17
2014–15	50,783	1,672.0	184.0	306.0	4,203.3	12.15
2015–16	51,064	1,610.9	178.1	302.7	4,217.8	12.11
2016–17	51,361	1,546.1	172.2	299.9	4,233.7	12.09
2017–18	51,609	1,513.7	168.0	298.9	4,243.0	12.06
2018–19	51,834	1,489.6	164.5	298.4	4,249.3	12.02
2019–20	52,053	1,464.8	161.3	298.2	4,257.8	11.97

a. CO_2 emissions estimate assumed all of the carbon that is oxidised is converted to CO_2.

Source BTRE (2003)

Table 44. Reference case rigid truck vehicle emission projections, by emission type, 2001–2020 (kt)

	Greenhouse	Air Pollutants				
Year	**$CO_{2\text{-}e}$[a]**	**CO**	**NO_x**	**VOCs**	**SO_x**	**PM**
2000–01	4,997	57.7	54.3	16.3	2,642.6	4.93
2001–02	5,127	55.6	54.5	15.8	2,420.8	4.94
2002–03	5,152	52.6	53.4	14.8	1,528.9	4.76
2003–04	5,276	50.5	53.3	14.2	945.4	4.63
2004–05	5,281	47.8	52.0	13.5	633.8	4.41
2005–06	5,321	44.9	50.9	12.8	165.8	4.23
2006–07	5,375	43.0	49.9	12.2	166.8	4.14
2007–08	5,402	41.0	48.6	11.7	166.9	4.02
2008–09	5,437	39.5	47.0	11.3	102.8	3.90
2009–10	5,501	38.6	45.4	10.9	38.2	3.81
2010–11	5,543	37.6	43.7	10.6	38.4	3.70
2011–12	5,579	36.9	42.0	10.2	38.7	3.60
2012–13	5,612	36.2	40.4	10.0	39.0	3.50
2013–14	5,643	35.6	38.9	9.7	39.3	3.42
2014–15	5,669	35.1	37.4	9.5	39.6	3.34
2015–16	5,700	34.6	36.0	9.3	39.9	3.27
2016–17	5,755	34.3	34.9	9.0	40.4	3.20
2017–18	5,771	33.9	33.7	8.9	40.6	3.13
2018–19	5,778	33.7	32.6	8.7	40.8	3.07
2019–20	5,778	33.5	31.6	8.5	41.0	2.99

a. CO_2 emissions estimate assumed all of the carbon that is oxidised is converted to CO_2.
Sources BTRE (2003)

11.2.6 Upstream (Pre-Combustion) Emissions

Estimates of total upstream emissions are not provided for either the reference case or the case where total biofuel consumption obtains 350 ML in 2010. Only the change in upstream emissions, between the reference case and the 350 ML biofuel use case, is provided (see Section 11.4).

11.3 Scenario 2: Fuel Use and Vehicle Emissions with 350 ML Biofuels Use in 2010

Under the case of 350 ML biofuel use in road transport, an extra 235 ML of biofuel consumption is required to obtain the required level of consumption. Based on the viability analysis it is assumed that an extra 30 ML is sourced from biodiesel and an additional 205 ML of ethanol (60 ML produced from C molasses and 145 ML from cereal grains—a mix of sorghum and feedwheat). The extra ethanol and biodiesel is assumed to displace an energy equivalent amount of gasoline and diesel fuel consumption, respectively. (For simplicity, it is assumed that the final price of fuels containing ethanol and biodiesel are not affected, so that total energy consumption remains as in the reference case.) As stated in Section 11.2, the study has assumed that all the additional ethanol would be consumed as E10 in passenger cars, and all additional biodiesel supply would be used as BD100 in rigid trucks. Some analysis of the sensitivity of the results to these assumptions is provided in chapter 13.

Table 45 lists the fuel consumption projections for passenger cars for the case where 350 ML of biofuels are consumed in 2010. Total energy required to fuel the passenger motor vehicle fleet is preserved; the extra ethanol consumption displaces an energy equivalent amount of ULP and PULP consumption. Table 46 lists the fuel consumption projections for rigid trucks under the case where 350 ML of biofuels are consumed in 2010.

Table 45. Passenger motor vehicle fuel consumption projections, 350 ML biofuels consumption case, by fuel type, 2000–2020 (million litres)

Year	LP/LRP	ULP	PULP	Total Petrol	ADO	LPG	NG[a] (Petrol equiv.)	Ethanol	Total
2000–01	2,932.3	11,875.2	774.6	15,582.0	650.0	1,450.0	9.0	40.0	17,731.0
2001–02	2,456.0	12,592.5	871.4	15,919.8	670.0	1,400.0	10.0	45.0	18,044.8
2002–03	2,081.4	13,406.0	980.3	16,467.7	680.0	1,450.0	11.9	50.0	18,659.6
2003–04	1,756.3	14,124.5	1,102.8	16,983.6	691.2	1,478.0	14.1	64.3	19,231.2
2004–05	1,464.5	14,749.5	1,240.7	17,454.6	702.7	1,506.6	16.7	82.6	19,763.3
2005–06	1,208.5	15,048.3	1,395.8	17,652.6	714.3	1,535.8	19.9	106.2	20,028.7
2006–07	1,018.3	15,260.2	1,570.3	17,848.7	726.1	1,565.4	23.6	136.5	20,300.3
2007–08	855.1	15,405.9	1,766.5	18,027.5	738.1	1,595.7	28.0	175.5	20,564.8
2008–09	727.2	15,465.3	1,987.4	18,179.8	750.3	1,626.6	33.2	225.6	20,815.5
2009–10	618.0	15,460.2	2,235.8	18,313.9	762.7	1,658.0	39.5	290.0	21,064.1
2010–11	472.0	15,459.2	2,515.3	18,446.4	775.3	1,690.1	46.9	290.0	21,248.7
2011–12	355.7	15,384.1	2,829.7	18,569.5	788.1	1,722.8	55.6	290.0	21,426.0
2012–13	263.8	15,223.3	3,183.4	18,670.5	801.1	1,756.1	66.0	290.0	21,583.8
2013–14	191.6	14,991.1	3,581.3	18,764.0	814.4	1,790.0	78.4	290.0	21,736.9
2014–15	135.4	14,677.9	4,028.9	18,842.3	827.9	1,824.7	93.1	290.0	21,877.9
2015–16	114.8	14,252.0	4,532.6	18,899.4	841.5	1,859.9	110.5	290.0	22,001.4
2016–17	97.6	13,765.4	5,099.1	18,962.1	855.5	1,895.9	131.2	290.0	22,134.7
2017–18	83.1	13,174.5	5,736.5	18,994.1	869.6	1,932.6	155.8	290.0	22,242.0
2018–19	70.9	12,487.0	6,453.6	19,011.5	884.0	1,969.9	185.0	290.0	22,340.4
2019–20	60.8	11,532.1	7,446.9	19,039.8	900.0	1,976.3	219.6	290.0	22,425.6

Notes LP/LRP = leaded petrol / lead replacement petrol (all sales in this category after 2001 are LRP).
ULP/PULP = premium unleaded petrol / premium unleaded petrol.
ADO – automotive diesel oil.
LPG – liquefied petroleum gas.
NG – natural gas.
For simplicity, total motor vehicle sales of ethanol (as E10) have been allocated to car fuel use.

Source BTRE (2003) and BTRE estimates.

Table 46. Rigid truck fuel consumption projections, 350 ML biofuels consumption case, by fuel type, 2000–2020 (million litres)

Year	Petrol	ADO	LPG	NG[a] (petrol eq.)	Biodiesel (BD100)	Total
2000–01	76.1	1,754.2	60.0	0.4	5.0	1,895.8
2001–02	64.1	1,809.9	65.0	0.7	5.0	1,944.7
2002–03	54.6	1,824.3	70.0	1.0	5.0	1,954.9
2003–04	47.1	1,873.5	75.0	1.3	10.0	2,006.9
2004–05	41.2	1,879.9	75.8	1.4	15.0	2,013.2
2005–06	31.0	1,903.0	76.5	1.4	20.0	2,032.0
2006–07	26.5	1,921.7	77.3	1.5	30.0	2,057.0
2007–08	22.0	1,930.5	78.0	1.6	40.0	2,072.2
2008–09	19.5	1,936.2	78.8	1.7	50.0	2,086.3
2009–10	19.0	1,950.9	79.6	1.9	60.0	2,111.3
2010–11	18.5	1,966.5	80.4	2.0	60.0	2,127.4
2011–12	19.0	1,978.7	81.2	2.2	60.0	2,141.1
2012–13	19.5	1,989.8	82.0	2.4	60.0	2,153.7
2013–14	20.0	2,000.2	82.8	2.6	60.0	2,165.7
2014–15	20.5	2,008.7	83.7	3.4	60.0	2,176.4
2015–16	21.0	2,018.4	84.5	4.5	60.0	2,188.4
2016–17	21.5	2,037.2	85.4	5.8	60.0	2,209.8
2017–18	22.0	2,041.1	86.2	7.6	60.0	2,216.9
2018–19	22.5	2,041.2	87.1	9.8	60.0	2,220.6
2019–20	23.0	2,038.2	87.9	12.8	60.0	2,221.9

Notes ADO – automotive diesel oil.
LPG – liquefied petroleum gas.
NG – natural gas.
For simplicity, total motor vehicle sales of ethanol (as E10) have been allocated to car fuel use.

Source BTRE (2003b) and BTRE estimates.

11.3.1 Vehicle Emissions for 350 ML Biofuel Consumption

Total passenger vehicle emissions for the case where there is an additional 205 ML of ethanol consumption in 2010 are listed in Table 47.

Total rigid truck emissions under the case where there is an additional 30 ML of biodiesel use by rigid trucks in 2010 are listed in Table 48.

Table 47. Passenger car vehicle emission projections, 350 ML biofuels consumption case, by emission type, 2001–2020 (kt)

	Greenhouse	Pollutants					
Year	**CO_2-e**	**CO**	**NOx**	**VOCs**	**SOx**	**PM**	**Lead**
2000–01	41,080	2,596.9	252.4	393.2	5,226.3	10.60	0.143
2001–02	41,929	2,508.3	249.9	380.9	4,977.7	10.74	0.023
2002–03	43,406	2,470.2	250.7	376.9	4,731.2	10.97	0.023
2003–04	44,794	2,429.6	250.5	375.8	4,562.4	11.23	0.021
2004–05	46,032	2,369.7	248.7	370.1	4,030.3	11.44	0.021
2005–06	46,607	2,284.5	242.9	360.4	3,899.8	11.49	0.020
2006–07	47,172	2,201.9	237.0	351.6	3,943.7	11.65	0.019
2007–08	47,701	2,119.7	231.0	347.3	3,983.8	11.80	0.018
2008–09	48,175	2,041.0	225.1	340.5	4,018.2	11.94	0.018
2009–10	48,612	1,960.0	219.1	334.5	4,049.3	12.05	0.017
2010–11	49,041	1,892.9	212.5	328.0	4,079.7	12.11	0.017
2011–12	49,448	1,824.0	205.3	322.2	4,108.1	12.15	0.017
2012–13	49,806	1,755.8	198.1	316.9	4,131.9	12.17	0.017
2013–14	50,154	1,690.5	191.3	312.3	4,154.0	12.17	0.017
2014–15	50,472	1,627.4	184.9	308.4	4,173.0	12.15	0.017
2015–16	50,753	1,568.1	178.9	305.1	4,187.5	12.11	0.017
2016–17	51,050	1,505.2	173.1	302.3	4,203.3	12.09	0.017
2017–18	51,297	1,473.8	168.8	301.3	4,212.7	12.06	0.016
2018–19	51,522	1,450.4	165.3	300.9	4,219.0	12.02	0.016
2019–20	51,741	1,426.3	162.1	300.7	4,227.4	11.97	0.016

Table 48. Rigid truck vehicle emission projections, 350 ML biofuels consumption case, by emission type, 2001–2020 (kt)

	Greenhouse	Pollutants					
Year	**CO_2-e**	**CO**	**NOx**	**VOCs**	**SOx**	**PM**	**Lead**
2000–01	4,997	57.7	54.3	16.3	2,642.6	4.93	2.788
2001–02	5,127	55.6	54.5	15.8	2,420.8	4.94	0.209
2002–03	5,152	52.6	53.4	14.8	1,528.9	4.76	0.170
2003–04	5,276	50.5	53.3	14.2	945.4	4.63	0.115
2004–05	5,281	47.8	52.0	13.5	633.8	4.41	0.096
2005–06	5,321	44.9	50.9	12.8	165.8	4.23	0.064
2006–07	5,362	43.0	49.9	12.2	166.4	4.13	0.041
2007–08	5,377	40.9	48.6	11.7	166.2	4.02	0.030
2008–09	5,386	39.3	47.0	11.2	101.9	3.89	0.024
2009–10	5,426	38.4	45.6	10.9	37.8	3.80	0.022
2010–11	5,469	37.4	43.8	10.5	38.0	3.69	0.020
2011–12	5,504	36.7	42.1	10.2	38.3	3.59	0.021
2012–13	5,537	36.0	40.5	10.0	38.6	3.50	0.021
2013–14	5,568	35.4	39.0	9.7	38.9	3.41	0.022
2014–15	5,596	34.9	37.6	9.5	39.2	3.33	0.022
2015–16	5,626	34.4	36.1	9.3	39.5	3.26	0.023
2016–17	5,682	34.1	35.0	9.0	40.0	3.20	0.023
2017–18	5,699	33.8	33.8	8.8	40.2	3.13	0.024
2018–19	5,706	33.5	32.7	8.7	40.4	3.06	0.025
2019–20	5,706	33.3	31.7	8.5	40.5	2.98	0.025

11.4 Impact on Total Emissions of Achieving 350 ML Biofuels Consumption in 2010

This section presents estimates of the change in vehicle emissions as a result of the increase in biofuels consumption between the reference case and consumption of 350 ML of biofuels in 2010, assuming all extra ethanol is consumed as E10 in passenger cars and all extra biodiesel use is consumed as BD100 by rigid trucks. Changes in tailpipe and upstream emissions are presented separately for ethanol and biodiesel. The change in CO_2, CH_4 and N_2O emissions are not presented separately, but combined as total greenhouse (CO_2-equivalent emissions). Annual estimates of the change in emissions are provided in Appendix IX.

11.4.1 Tailpipe Emissions

Table 49 shows the change in projected tailpipe emissions from consumption of an extra 205 ML of ethanol in 2009–2010, relative to the reference case level of ethanol consumption (311.3 thousands). Increased ethanol consumption results in increased tailpipe output of CO_2 and NO_x (1,087 tonnes in 2009-10), but reduced output of CO (55,500 tonnes in 2009-2010) and PM (approximately one tonne in 2009-2010). Total greenhouse emissions (CO_2-e) from passenger cars would decline by 311,300 tonnes in 2009-2010 because the CO_2 from the ethanol component is from bio-sources. Because the sulfur content of ULP/PULP is assumed to remain unchanged from 2010, substitution of ethanol for ULP/PULP results in a fixed reduction in total SO_x emissions (not shown in Table 49) from that date.

As a result of the extra ethanol use, total VOC emissions initially decline and then increase over the projection horizon. This result is attributable to the different effect of E10 on exhaust and evaporative VOC emissions. VOC exhaust emissions are projected to fall by 1900 tonnes in 2010, as a result of the substitution of E10 for petrol. However, evaporative VOC emissions are projected to increase by 4,260 tonnes in 2010, due to the higher volatility of ethanol blend fuels. The net impact is a decline in total VOC emissions early in the projection horizon. In 2009-2010, total VOC emissions would increase by 2,360 tonnes due to the increase in E10 consumption.

Table 49. Change in projected tailpipe emissions from increased ethanol consumption, 2002–2003 to 2019–2020 (tonnes)

	Greenhouse	Pollutants				
Year	CO_2-e	CO	NO_x	VOC exhaust	VOC evaporative	PM
2002–03	0	0	0	0	0	0
2004–05	-34,942	-7768	147.5	-270.4	535.6	-0.129
2009–10	-311,313	-55531	1086.6	-1900.6	4260.4	-1.029
2014–15	-311,355	-44588	886.3	-1581.7	3977.1	-0.913
2019–20	-311,367	-38469	762.6	-1441.5	3932.3	-0.819

Table 50 shows the change in projected tailpipe emissions due to increased biodiesel consumption, which is all assumed to occur in urban areas. Because it is assumed that there is no extra biodiesel consumption in the 350 ML case until 2006-2007, there is no change in emissions until after that date. Substitution of BD100 waste oil for diesel in the 350 ML biofuels case results in reduction in total greenhouse emissions (of approximately 75,000 tonnes in 2009-2010) and a reduction in pollutant emissions for all major pollutants but NO_x. Particle emissions are estimated to decrease by 5.8 tonnes in 2009-2010. At that time, it is assumed that XLSD is the standard automotive diesel fuel in use, which results in a smaller reduction in PM emissions from substitution of BD100 for diesel than if LSD or ULSD were the standard diesel fuel.

Table 50. Change in projected tailpipe emissions from increased biodiesel consumption, 2002–2003 to 2019–2020 (tonnes)

	Greenhouse	Pollutants			
Year	**CO_2-e**	**CO**	**NO_x**	**VOCs[a]**	**PM**
2002–03	0.0	0.0	0.0	0.0	0.0
2004–05	0.0	0.0	0.0	0.0	0.0
2009–10	-75,011	-207.6	173.3	-18.6	-5.85
2014–15	-73,823	-193.5	138.5	-15.7	-5.05
2019–20	-72,116	-194.3	115.0	-13.6	-4.34

[a] Exhaust plus evaporative VOC emissions.

Table 51 shows the net change in tailpipe emissions resulting from 350 ML of biofuels use by 2010. Total greenhouse emissions from vehicle use are projected to decline by 386,000 tonnes in 2009–2010 as a result of increased ethanol and biodiesel consumption. CO and PM emissions also are projected to decline, by 55,800 tonnes and 6.9 tonnes, respectively. NO_x and total VOC emissions are projected to increase, by 1,260 tonnes and 2,340 tonnes in 2009-2010.

Table 51. Change in projected tailpipe emissions from increased biofuel consumption, 2001–2020 (tonnes)

	Greenhouse		Pollutants			
Year	**CO_2**	**CO**	**NO_x**	**VOCs[a] exhaust**	**VOCs Total**	**PM**
2002–03	0.0	0.0	0.0	0.0	0.0	0.0
2004–05	-34,941	-7,768	148	-270	265.2	-0.13
2009–10	-386,324	-55,738	1,260	-1,919.2	2,341.2	-6.88
2014–15	-385,178	-44,782	1,025	-1,597	2,379.7	-5.86
2019–20	-383,482	-38,663	878	-1,455	2,477.2	-5.16

[a] Exhaust plus evaporative VOC emissions.

Relative to total projected road transport source emissions, the increase in total biofuel use would reduce total greenhouse emissions from road transport sources by approximately 0.7% in 2009–2010. Of total road transport source pollutant emissions, CO and PM emissions would fall by 2.7% and 0.04% in 2009–2010, and total NO_x and VOC emissions would increase by 0.5 and 0.7%, respectively. Table 52 shows the change in tailpipe emissions, as a share of total road transport sector emissions, resulting from supply of 350 ML of biofuels in 2010. Total road transport sector greenhouse and criteria pollutant emissions are listed in Appendix IX.

Table 52. Percentage change in projected tailpipe emissions from 350 ML biofuel consumption, 2002–2003 to 2019–2020 (%)

	Greenhouse	Pollutants			
Year	**CO_2-e.**	**CO**	**NO_x**	**VOCs**	**PM**
2002–03	0	0	0	0	0
2004–05	-0.050	-0.25	0.03	0.06	0.00
2009–10	-0.511	-2.12	0.30	0.56	-0.03
2014–15	-0.480	-2.02	0.28	0.61	-0.03
2019–20	-0.458	-1.97	0.27	0.66	-0.02

11.4.2 Upstream Emissions

Upstream emissions include only CO_2 emissions arising from production of biofuels. In accordance with the Kyoto Protocol convention, CO_2 emissions taken-up in growing the feedstock are accounted for at the combustion stage. Chapter 4 provided a discussion of the greenhouse accounting conventions adopted in this study.

The change in biofuels production required to obtain 350 ML of biofuels consumption entails an additional 205 ML of ethanol, from various sources, and 30 ML of biodiesel. In calculating the impact on tailpipe emissions, it was assumed that the additional ethanol and biodiesel displaces an energy equivalent amount of petrol and diesel, respectively. In computing the change in upstream emissions, it is assumed that domestic production of petrol and diesel is reduced by the amount petrol and diesel consumption displaced by the extra biofuel use.

Ethanol

Table 53 shows the amount of additional ethanol production, by feedstock source, required to obtain 350 ML biofuels consumption, and the amount of petrol that it would replace. The additional 205 ML of ethanol required by 2010 is assumed to be based on a mix of C molasses and cereal grain feedstocks. It has been assumed that an additional 60ML of ethanol is produced from C molasses and 145ML from cereal grains. For convenience, a graduated increase in additional future ethanol production is assumed, with the different feedstocks coming on stream at different times and rates. The order in which the different feedstocks are assumed to come on stream affects the change in emissions up to 2009, but does not affect the results in 2010, when all of the additional ethanol is assumed to be taken up.

Table 53. Change in ethanol and petrol production between the reference case and 350 ML biofuels consumption case (ML)

	Ethanol				
Year	Waste starch (existing capacity)	C molasses (co-generated energy) (existing capacity)	Sorghum (new capacity)	Wheat (new capacity)	ULP / PULP
2000–01	0	0	0	0	0
2001–02	0	0	0	0	0
2002–03	0	0	0	0	0
2003–04	0	10.3	0	0	-7.1
2004–05	0	24.4	0	0	-16.7
2005–06	0	43.4	0	0	-29.7
2006–07	0	60.0	8.8	0	-47.1
2007–08	0	60.0	42.5	0	-70.1
2008–09	0	60.0	80.0	6.8	-100.4
2009–10	0	60.0	80.0	65.0	-140.3
2010–11	0	60.0	80.0	65.0	-140.3
2011–12	0	60.0	80.0	65.0	-140.3
2012–13	0	60.0	80.0	65.0	-140.3
2013–14	0	60.0	80.0	65.0	-140.3
2014–15	0	60.0	80.0	65.0	-140.3
2015–16	0	60.0	80.0	65.0	-140.3
2016–17	0	60.0	80.0	65.0	-140.3
2017–18	0	60.0	80.0	65.0	-140.3
2018–19	0	60.0	80.0	65.0	-140.3
2019–20	0	60.0	80.0	65.0	-140.3

In calculating the change in upstream emissions resulting from the increased supply of biofuels, the study has applied volumetric upstream emissions rates, specified in terms of grams per litre of fuel produced, to the change in total biofuel and conventional fuel production.

Table 56 presents the volumetric upstream emission rates for ethanol. These rates are based on the per kilometre emission rates discussed in Chapter 10.

CO_2 emissions rates for production of ethanol from waste starch, sorghum and wheat are as much as three times the CO_2 emissions per litre of ULP. N_2O emissions rates are higher for ethanol, than ULP, for all feedstocks other than sorghum. Pollutant emission rates, for all but NMVOCs, also are much higher for ethanol production than for ULP production. Again, this is probably due to assumptions about the source of non-renewable energy used in production.

Table 54. Upstream emission rates – ethanol and ULP (grams per litre)

	Greenhouse			Pollutants			
Fuel	**CO_2**	**CH_4**	**N_2O**	**CO**	**NO_x**	**NMVOCs**	**PM**[a]
Ethanol – waste starch	758.4	0.809	0.00981	1.002	2.266	0.120	0.4085
Ethanol – molasses (cogeneration energy)	364.3	-0.07279	0.2508	11.58	1.692	0.313	0.02109
Ethanol – sorghum	979.6	1.524	-0.2151	0.909	2.455	0.140	0.3952
Ethanol – wheat	1,034.1	1.471	0.5778	15.86	5.669	1.249	0.4646
ULP	332.20	3.133	0.00195	0.5354	2.810	1.105	0.08452

[a] PM emissions for ethanol based on non-urban sites. PM emissions for petrol from urban sites.

Note Ethanol denotes pure ethanol
ULP – Unleaded petrol.

The upstream emission rates listed in Table 54 were applied to the change in total ethanol and petrol production to compute the impact on upstream emissions, listed in Table 55 and Table 57. The separate listing of the upstream emissions from ethanol and petrol are maintained in order to compute the differential health impact between urban and non-urban areas. All of the ethanol production is assumed to take place in non-urban or provincial urban centres, whereas petrol production occurs in metropolitan centres in Australia.

Table 55. Change in upstream emissions from additional ethanol production, 2001–2020 (tonnes)

	Greenhouse			Pollutants			
Year	**CO_2**	**CH_4**	**N_2O**	**CO**	**NO_x**	**VOCs**	**PM**
2002–03	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2004–05	8,902.3	-1.8	6.1	283.0	41.3	7.6	0.5
2009–10	167,439.6	213.2	35.4	1,798.5	666.4	111.2	63.1
2014–15	167,439.6	213.2	35.4	1,798.5	666.4	111.2	63.1
2019–20	167,439.6	213.2	35.4	1,798.5	666.4	111.2	63.1

Table 56. Change in upstream emissions from reduction in domestic petrol production, 2001–2020 (tonnes)

	Greenhouse			Pollutants			
Year	**CO_2**	**CH_4**	**N_2O**	**CO**	**NO_x**	**VOCs**	**PM**
2002–03	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2004–05	-5,554.2	-52.4	0.0	-9.0	-47.0	-18.5	-1.41
2009–10	-46,595.4	-439.4	-0.3	-75.1	-394.1	-155.0	-11.86
2014–15	-46,595.4	-439.4	-0.3	-75.1	-394.1	-155.0	-11.86
2019–20	-46,595.4	-439.4	-0.3	-75.1	-394.1	-155.0	-11.86

As indicated in Table 57, the net change from increased ethanol production is an overall increase in total greenhouse emissions, as well as increases in most criteria pollutants, except for NMVOCs, which are projected to decline.

Table 57. Net change in upstream emissions from change in domestic ethanol and petrol production, 2001–2020 (tonnes)

	Greenhouse			Pollutants			
Year	CO_2	CH_4	N_2O	CO	NO_x	VOCs	PM
2002–03	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2004–05	3,348.1	-54.2	6.1	274.1	-5.6	-10.8	-0.90
2009–10	120,844.2	-226.3	35.1	1,723.4	272.2	-43.8	51.22
2014–15	120,844.2	-226.3	35.1	1,723.4	272.2	-43.8	51.22
2019–20	120,844.2	-226.3	35.1	1,723.4	272.2	-43.8	51.22

Biodiesel

Supply of 350 ML of biofuels also entails an assumed additional 30 ML of biodiesel consumption in 2010, produced from waste cooking oil. This additional biodiesel production displaces approximately 27 ML of conventional diesel. The substitution of biodiesel for diesel also results in changes in total upstream emissions. Table 58 shows the assumed upstream emissions rates derived for the current study. Biodiesel from waste oil produces significantly less greenhouse and pollutant emissions per litre than for LSD, ULSD and XLSD.

Table 58. Upstream emission rates – biodiesel and diesel (grams per litre)

	Greenhouse			Pollutants			
Fuel	CO_2	CH_4	N_2O	CO	NO_x	NMVOCs	PM[a]
Biodiesel – waste oil	251.2	0.4411	0.00131	0.4858	0.9487	0.1499	0.00669
Low sulfur diesel	429.8	3.688	0.00251	0.9548	3.440	1.397	0.05094
Ultra low sulfur diesel	484.6	3.669	0.00308	1.039	3.847	1.419	0.05303
Extra low sulfur diesel	543.5	3.696	0.00367	1.133	4.288	1.457	0.05572

[a] PM emissions for biodiesel based on non-urban sites. PM emissions for diesel from urban sites.

Table 59 shows the additional upstream emissions resulting from the extra biodiesel production assumed under the 350 ML biofuel supply case, for the period 2002-2003 to 2019-2020. Table 60 shows the projected reduction in upstream emissions from the reduction in domestic diesel production resulting from the switch from diesel to biodiesel. In calculating the reduction in upstream emissions from domestic diesel production, it is assumed that all automotive diesel produced from 2002–03 is LSD, from 2005–06 all automotive diesel is ULSD and from 2009–10 all automotive diesel is XLSD (i.e. sulfur content of 10 ppm).

Table 59. Change in upstream emissions from additional biodiesel production, 2001–2020 (tonnes)

	Greenhouse			Pollutants			
Year	CO_2	CH_4	N_2O	CO	NO_x	NMVOCs	PM
2002–03	0.0	0.0	0.0	0.0	0.0	0.0	0.000
2004–05	0.0	0.0	0.0	0.0	0.0	0.0	0.000
2009–10	7,536.0	13.2	0.039	14.6	28.5	4.50	0.235
2014–15	7,536.0	13.2	0.039	14.6	28.5	4.50	0.235
2019–20	7,536.0	13.2	0.039	14.6	28.5	4.50	0.235

Table 60. Change in upstream emissions from reduction in domestic diesel production, 2001–2020 (tonnes)

	Greenhouse			Pollutants			
Year	CO_2	N_2O	CH_4	CO	NO_x	NMVOCs	PM
2002–03	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2004–05	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2009–10	-14,638.9	-99.5	-0.1	-30.5	-115.5	-39.2	-1.50
2014–15	-14,638.9	-99.5	-0.1	-30.5	-115.5	-39.2	-1.50
2019–20	-14,638.9	-99.5	-0.1	-30.5	-115.5	-39.2	-1.50

Table 61 shows the net change in upstream emission from biodiesel. Replacement of diesel with biodiesel would result in net reductions in both greenhouse and pollutant emissions from upstream sources.

Table 61. Net change in upstream emissions from reduction in domestic biodiesel and diesel production, 2001–2020 (tonnes)

	Greenhouse			Pollutants			
Year	CO_2	N_2O	CH_4	CO	NO_x	NMVOCs	PM
2002–03	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2004–05	0.0	0.0	0.0	0.0	0.0	0.0	0.0
2009–10	-7,102.9	-86.3	-0.1	-15.9	-87.0	-34.7	-1.27
2014–15	-7,102.9	-86.3	-0.1	-15.9	-87.0	-34.7	-1.27
2019–20	-7,102.9	-86.3	-0.1	-15.9	-87.0	-34.7	-1.27

11.4.3 Combined Impact – Tailpipe and Upstream Emissions

Table 62 shows the impact on greenhouse gas and pollutant emissions of increasing total biofuels consumption from the reference case level of 115 ML to 350 ML in 2010.

The increased consumption of biofuels would result in reduced greenhouse emissions, of approximately 0.27 million tonnes in 2010. Most of the reduction in greenhouse emissions results from the reduced tailpipe emissions, estimated to decline by 0.39 million tonnes. This apparent decline in ‘tailpipe’ emissions is due to the convention of ignoring carbon emissions when the fuel, in this case ethanol and biodiesel, is produced from renewable sources. Upstream emissions, however, are projected to increase significantly, by 0.12 million tonnes, due to the more carbon intensive energy sources assumed in this study to be used in ethanol production.

Of the major criteria pollutants, it is estimated that the increase in biofuels consumption necessary to obtain 350 ML in 2010 would result in decreased CO emissions, but increases in NO_x, VOC and PM emissions. Most of the reductions in CO emissions are attributable to the lower CO emissions of E10 blend fuels relative to unleaded petrol. The projected increases in NO_x and total VOCs also arise primarily from the combustion of E10. Ethanol blends produce higher NO_x than petrol. The higher output of total VOCs is the net result of two effects: a reduction in exhaust emissions but an increase in evaporative emissions, due to the higher volatility of E10. Total particle emissions are also projected to increase, largely because of the large PM emissions arising from production of ethanol.

Table 62. Change in greenhouse and criteria pollutant emissions resulting from consumption of 350 ML of Biofuels in 2010 (tonnes)

	Greenhouse	Pollutants			
	CO_2-e	CO	NO_x	VOCs	PM
Ethanol					
Upstream	126,980	1,723.4	272.2	-43.8	51.22
Tailpipe	-311,313	-55,530.6	1,086.6	2,359.8	-1.03
Total	-184,333	-53,807.2	1,358.8	2,316.0	50.20
Biodiesel					
Upstream	-8,934	-15.9	-87.0	-34.7	-1.30
Tailpipe	-75018	-207.6	173.3	-18.6	-5.85
Total	-83,952	-223.5	86.3	-53.3	-7.10
Biofuels					
Upstream	118,046	1,707.5	185.2	-78.5	49.92
Tailpipe	-386,331	-55,738.2	1,259.9	2,341.1	-6.88
Total	-268,278	-54,030.7	1,445.1	2,262.6	43.10

Note: Figures may not add to total due to rounding.

12 HEALTH COST IMPACTS

Health impacts of air pollutants have been outlined in Chapters 8 and 10. This chapter presents estimates of the health cost impact of the change in total emissions resulting from 350 ML of biofuel consumption in 2010. The health costs will depend on both the size and location of any changes in emissions, and the unit health costs assumed for each pollutant. In estimating the impact on total health costs resulting from an increase in biofuels consumption, the study team has made assumptions about the location of production facilities and where consumption of biofuels is most likely to occur. The study has drawn on the most recent published estimates of the unit health costs of vehicular pollutant emissions.

With respect to greenhouse gases and climate change, the implications for human health include: heat-related illnesses, injury and death due to more frequent and severe extreme weather, increased risk of contagious diseases due to a wider range of insect vectors, water and food-borne diseases, cancer from exposure to dangerous levels of solar ultraviolet radiation.

Although the threats to health posed by climate change are widely discussed in the literature, and numerous national and supra-national institutions undertake the monitoring, quantification still remains a challenge. Valuation of reduction in greenhouse emissions is discussed in Chapter 15.

12.1 Biofuel Use Assumptions

At present, most of the ethanol used in transport in Australia is generally consumed in the Greater Sydney metropolitan area (taking in Newcastle–Sydney–Wollongong), close to the location of the majority of current production (Manildra's ethanol refinery near Nowra). For the reference case it is assumed that all of the 85 ML of ethanol produced in 2010 will be consumed in major metropolitan areas. Likewise, as all of the 30 ML biodiesel production in 2010 under the reference case is assumed to be produced from waste oil collected from metropolitan centres, all of the transport use of that biodiesel is assumed to be undertaken in metropolitan areas.

The study has assumed that consumption of 350 ML of biofuels in 2010, would entail an additional 205 ML of ethanol—60 ML from C molasses and 145 ML from grain feedstocks (80 ML from sorghum and 65 ML from wheat)—and 30 ML of biodiesel, sourced from waste cooking oil. In estimating the health impact of upstream emissions, it is assumed that the extra ethanol production would take place in three separate rural locations: southwest Queensland (using cereal grain feedstock) and Northern Queensland (molasses feedstock) and northern NSW (cereal grain feedstock). The extra biodiesel production is, for the purposes of this analysis, assumed to occur on the fringe of a large metropolitan areas, within reasonable proximity of the raw feedstock.

In calculating the health impact of emissions produced from the additional transport use of biofuels, it is assumed that most of the transport use occurs in major metropolitan areas. It is assumed that ethanol produced from new grain-based sources would have to be transported to the petrol refineries in Sydney and Brisbane, for blending with petrol. It assumed that the E10 blend would be then sold in the major metropolitan centres of Greater Sydney (Newcastle–Sydney–Wollongong) and South East Queensland (Gold Coast–Brisbane–Sunshine Coast). Where the ethanol sourced from C molasses will be consumed will depend on whether the ethanol has to be taken to the refineries (in the major capitals) for blending or whether it can be blended elsewhere. For the current analysis it has been assumed that all of the extra ethanol produced from C molasses would be used primarily in markets in Central and Northern Queensland.[22] The consequent health impact of tailpipe emissions from the ethanol produced from C molasses will be less pronounced because population densities are much lower in Central and Northern Queensland.

The additional biodiesel supply is assumed to be consumed in metropolitan areas (refer to Chapter 11 for discussion), providing a more pronounced health impact due to the reduction in PM emissions.

Table 63 shows the change in emissions in urban and rural areas resulting from the increase in biofuels supply and use, based on the aforementioned assumptions about the location of production and consumption. The figures show that most of the reduction in CO emissions occurs in urban areas. Much of the overall increase in total NO_x emissions occurs in rural areas due to the increase in NO_x arising from ethanol production. Most of the increase in total VOCs occurs in urban areas, due to increased evaporative emissions from E10 blend fuels. Particle emissions are projected to increase overall, due to the increased particle emissions from ethanol production, most of this increase in particle emissions occurs in rural areas. The reduction in production of petrol and diesel and substitution of biodiesel for diesel, which is assumed to take place in urban areas, results in a reduction in total urban PM emissions.

12.2 Unit Health Costs of Vehicle Emissions

Epidemiological studies have shown a link between concentrations of toxic substances in urban air sheds and morbidity and mortality rates amongst residents. The common (‘criteria’) ambient air pollutants, which include CO, NO_x, O_3, SO_x, PM and lead, are associated with a large range of non-cancerous health effects, from temporary impairment of lung function to increases in mortality. Toxic substances, such as benzene, 1,3-butadiene and polycyclic aromatic hydrocarbons also impact upon human health. Motor vehicle use in urban areas makes a significant contribution to the concentration of criteria and toxic pollutants in these airsheds.

There is an increasing volume of literature devoted to estimating the economic cost of increased morbidity and mortality due to increased concentrations of the more common pollutants. Estimates incorporate the monetary value of loss of life (mortality) and lost quality of life (morbidity), as well as health system costs. Empirical studies exhibit a considerable range of variation, with more recent studies generally attributing a higher cost to pollutant emissions, and particularly PM emissions, than earlier studies. This may reflect more recent scientific research, which measures the longer-term rather than shorter-term impact of air pollutants on health (see, for example, Amoako *et al.*, 2003), as well as other findings about the health impact of PM10 and PM2.5 (Fisher *et al.*, 2002).

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[22] The Survey of Motor Vehicle Use (ABS, 2003) reports a total of over 16,000 million kilometres travelled by passenger motor vehicles in Queensland outside Brisbane, which would imply around 1.4 to 1.6 billion litres of petrol consumption—sufficient to blend with the 60 ML of ethanol that would produced from C molasses.

Table 63. Change in greenhouse and criteria pollutant emissions resulting from consumption of 350 ML of Biofuels in 2010, by location (tonnes)

	Urban areas				**Rural areas**				***All areas***			
	CO	**NO_x**	**VOCs total[a]**	**PM**	**CO**	**NO_x**	**VOCs total[a]**	**PM**	**CO**	**NO_x**	**VOCs total[a]**	**PM**
Ethanol												
Upstream	-75.1	-394.1	-155.0	-11.9	1,798.5	666.4	111.2	63.1	1,723.4	272.2	-43.8	51.2
Tailpipe	-49,977.6	977.9	2,123.9	-0.9	-5,553.1	108.7	235.9	-0.1	-55,530.6	1,086.6	2,359.8	-1.0
Total	-50,052.7	583.8	1,968.9	-12.8	-3,754.5	775.0	347.1	63.0	-53,807.2	1,358.8	2,316.0	50.2
Biodiesel												
Upstream	-15.9	-87.0	-34.7	-1.3	0.0	0.0	0.0	0.0	-15.9	-87.0	-34.7	-1.3
Tailpipe	-207.6	173.3	-18.6	-5.9	0.0	0.0	0.0	0.0	-207.6	173.3	-18.6	-5.9
Total	-223.6	86.3	-53.4	-7.2	0.0	0.0	0.0	0.0	-223.6	86.3	-53.4	-7.2
Biofuels												
Upstream	-91.0	-481.2	-189.7	-13.2	1,798.5	666.4	111.2	63.1	1,707.5	185.2	-78.5	49.9
Tailpipe	-50,185.2	1,151.2	2,105.3	-6.8	-5,553.1	108.7	235.9	-0.1	-55,738.2	1,259.9	2,341.1	-6.9
Total	-50,276.2	670.1	1,915.6	-20.0	-3,754.5	775.0	347.1	63.0	-54,030.7	1,445.1	2,262.6	43.1

[a] exhaust and evaporative emissions

Table 64. Assumed unit health costs for pollutant emissions ($A / tonne)

Emission	Band 1	Band 2	Band 3	Band 4
NO_x	1,750	1,750	260	0
CO	3	0.8	0.8	0
NMVOCs	850	880	180	0
SO_x	11,380	4,380	2,800	50
PM	341,650	93,180	93,180	1,240

na not available.
Note Band 1 = Inner areas of larger capital cities (Sydney, Melbourne, Brisbane, Adelaide and Perth).
Band 2 = Outer areas of larger capital cities.
Band 3 = Other urban areas, including other capital cities (Canberra, Hobart and Darwin) and other urban areas.
Band 4 = Non-urban areas.
Source Watkiss (2002).

Against this background and to avoid any undue complexity, this study adopts health cost estimates derived for Australian conditions by Watkiss (2002), presented in Table 64. Watkiss' unit health costs are based on European health cost estimates (derived as part of the ExternE project, http://externe.jrc.es/), adjusted for the demographic characteristics of Australian urban areas. Watkiss provides separate unit health cost estimates that vary according to population density ('Bands' 1 to 4). Watkiss' unit health cost estimates for PM, although of the same order of magnitude as some other Australian studies, notably Beer (2002), Coffey (2003) and Amoako *et al.* (2003) are at the upper end of the range. For example, PM emissions, which typically have the highest unit costs, around $A100,000–300,000 per tonne in built up urban areas, are estimated by Watkiss to be approximately $A341,000 per tonne in the inner areas of major Australian metropolitan centres. (Appendix VIII provides a brief review of estimates of the unit health costs of emissions).

In computing the total health costs resulting from obtaining 350 ML of biofuels use, triangular distributions were imposed on Watkiss' (2002) unit cost estimates for each location (Appendix X). This procedure slightly alters the mean (average) unit cost estimates for each location. The mean unit health costs estimates used to estimate the total costs are shown in Table 65.

Table 65. Assumed average 'low' and 'high' unit health costs for 'criteria' pollutant emissions ($A / tonne)

Emission	Band 1	Band 2	Band 3	Band 4
CO	2.3	1.5	0.5	0.3
NO_x	1,253.3	756.7	173.3	86.7
NMVOCs	643.3	411.7	120.0	60.0
PM	258,827	176,003	62,533	31,887

Note Band 1 = Inner areas of larger capital cities (Sydney, Melbourne, Brisbane, Adelaide and Perth).
Band 2 = Outer areas of larger capital cities.
Band 3 = Other urban areas, including other capital cities (Canberra, Hobart and Darwin) and other urban areas.
Band 4 = Non-urban areas.
Source Watkiss (2002).

12.3 Health Cost Impact

The health cost impacts include the cost of additional upstream emissions resulting from production of biofuels, the benefits from reduced upstream emissions from the reduction in the production of ULP/PULP and diesel in metropolitan areas, and the change in the health costs arising due to the substitution of ethanol blend fuel and biodiesel for ULP/PULP and diesel. Monetary amounts are expressed in 2003 dollars.

For ethanol it is assumed that, in line with the assumptions outlined in Chapter 11, 145 ML of the additional 205 ML of ethanol is consumed in major metropolitan centres (Band 1 and Band 2 areas in Watkiss, 2002) and 60 ML is consumed in non-urban areas, e.g. northern Queensland, (Band 3 and

Band 4 areas). Using these assumptions, Table 66 provides estimates of the impact on health costs from the assumed increase in ethanol use.

The results show that the change in total exbodied pollutant emissions from an extra 205 ML of ethanol use would result in savings in total health costs of approximately $1.8 million in 2010, an average saving of 0.9c/L. Most of the savings in health costs are attributable to reduced PM emissions in urban areas, primarily due to the assumed reduced production of petrol at refineries based in metropolitan areas. Total VOC emissions are projected to increase due to increased evaporative emissions as a result of increased use of E10. Exhaust VOC emissions, however, would decline as a result of increased ethanol use in transport fuels. Because of a lack of knowledge about the detailed composition of evaporative emissions from E10, in terms of both ozone precursors and air toxics, the study has not applied a cost to the increase in evaporative emissions. Hence the health cost impact for tailpipe VOC emissions from ethanol relates only to the exhaust VOC component.

Table 66. Health cost impact of change in pollutant emissions resulting from increased consumption of biofuels in 2010

Fuel type source	Change in biofuels	Change in emissions (tonnes)				Cost	
		CO	NO_x	VOCs	PM	Total	Average
	ML					$ million	c/L
Ethanol							
Upstream	205	1,723.4	272.2	-43.8	51.22	-1.6	-0.8
Tailpipe	205	-55,530.6	1,086.6	2,359.8	-1.03	-0.2	-0.1
Total	205	-53,807.2	1,358.8	2,316.0	50.20	-1.8	-0.9
Biodiesel							
Upstream	30	-15.9	-87.0	-34.7	-1.30	-0.5	-1.7
Tailpipe	30	-207.6	173.3	-18.6	-5.85	-1.0	-3.5
Total	30	-223.5	86.3	-53.3	-7.10	-1.5	-5.2
Biofuels							
Upstream	235	1,707.5	185.2	-78.5	49.92	-2.1	-0.9
Tailpipe	235	-55,738.2	1,259.9	2,341.1	-6.88	-1.2	-0.5
Total	235	-54,030.7	1,445.1	2,262.6	43.04	-3.3	-1.4

The savings in health costs from an additional 30 ML of biodiesel consumption are estimated to be $1.5 million in 2010. This represents an average saving of around 5c/L of biodiesel. The savings in emissions due to the substitution of biodiesel for automotive diesel is much less in 2010 than in earlier years, because from 2010 it is assumed that XLSD will be the only form of automotive diesel available.

The total health impact of 350 ML of biofuel use in 2010 is then estimated to be $3.3m in 2010, an average health cost saving of 1.4c/L of additional biofuel use.

13 UNCERTAINTY AND SENSITIVITY ANALYSIS

The estimated impact of an increase in biofuel supply on greenhouse and pollutant emissions, and consequently health costs, are subject to a range of uncertainties. In particular, this study has had to make assumptions about the type and mix of feedstocks that would be used to produce the additional amount of biofuels, and where the biofuels would be consumed and produced. Additionally, vehicle emissions test results invariably exhibit a range of variation both between repeated tests of the same vehicle and between different vehicles. The measured variation in emission rates may be used to provide an indication of the significance of the estimated change in total emissions.

This chapter presents the results of uncertainty and sensitivity analysis of the estimated change in emissions and health costs presented in Chapters 11 and 12. Three types of analysis are presented here. Section 13.1 provides estimates of the impact on emissions and health costs of changes in some of the critical assumptions used in the analysis ('scenario' analysis). Section 2 presents estimates of the range of uncertainty associated with the estimates ('probabilistic uncertainty' analysis). The final section provides an indication of the relative sensitivity of total health costs to changes in the level of different pollutants.

13.1 Scenario Analysis

Analysis was undertaken to test the sensitivity of the emissions estimates to variation in some of the critical assumptions. The key assumptions tested were:

- the impact on emissions of sourcing all additional ethanol from C molasses using co-generated energy sources;
- the impact on upstream emissions of using gas rather than coal (and gas-fired electricity rather than coal fired electricity) in the production of ethanol;
- the impact on emissions of assuming the biodiesel is consumed as BD20 or BD5 blends rather than BD100; and
- the impact of assumptions about where the extra ethanol and biodiesel would be consumed.

13.1.1 Impact on Sourcing All Additional Ethanol from C Molasses Using Co-Generation

The emissions estimates presented in Chapter 11 ('main scenario') assumed the additional 205 ML of ethanol from new capacity would be supplied by 60 ML of ethanol produced from C molasses feedstocks and 145 ML from cereal grains. The level of whole grains in this mix was favoured on account of the slightly lower cost per litre (net of other by-product revenues) of ethanol from this source (Chapter 7). The economic viability of ethanol production from whole grains and C molasses is sufficiently similar to warrant consideration of alternative scenarios. This section estimates the impact on emissions and health costs of an alternative scenario whereby all additional ethanol production (205 ML) is sourced from C molasses, using cogeneration in sugar mills, based on year-round bagasse supply.

This scenario is likely to represent a 'best case' circumstance in terms of environmental benefits on account of this form of ethanol production having the highest greenhouse abatement potential and lowest PM emissions.

An implicit assumption in this scenario is that sugar mills would develop cost-effective C molasses storage facilities to enable year-round ethanol distillery operations and that these storage arrangements would mean that sufficient C molasses would be available (at the assumed average price of $50/tonne) to produce 205 ML of ethanol. The entire current C molasses supply could provide feedstock for approximately 300 ML of ethanol, so diverting an amount sufficient to produce 205 ML is technically feasible, although whether this is practically feasible given competing demands for C molasses is not clear. Year-round cogeneration also requires more comprehensive bagasse storage and handling facilities than are currently the norm in sugar mills, although these facilities are considered technically feasible and are starting to appear in sugar mills (e.g. northern NSW and North Queensland).

Table 67 shows the change in upstream, and total, emissions for the difference between the reference case and the 350 ML case (where the additional ethanol is produced from a combination of C molasses and grains), hereafter referred to as the 'main scenario' and for the case where all of the additional ethanol is produced from C molasses using co-generated energy sources. For the case where all the extra ethanol is produced from C molasses using co-generated energy, total CO_2-e emissions would fall by 277,000 tonnes in 2010, rather than 184,333 tonnes as under the main scenario. Total CO emissions would not fall by as much, while NO_x and total VOC emissions would increase by less. Total PM emissions from production and use of ethanol would fall by 8.5 tonnes in 2010. In the main scenario total PM emissions were projected to increase by 50 tonnes in 2010.

Table 67. Impact on emissions of sourcing all additional ethanol from C molasses using co-generated energy sources, in 2010

	Greenhouse		**Pollutants**			**Health costs**	
	CO_2-e	**CO**	**NO_x**	**VOCs**	**PM**	Total	Average
	(tonnes)		**(tonnes)**			($m)	(c/L)
Main scenario							
Upstream	126,980	1,723	272	-44	51.20	-1.6	-0.8
Tailpipe	-311,313	-55,531	1,087	2,360	-1.03	-0.2	-0.1
Total	**-184,333**	**-53,807**	**1,359**	**2,316**	**50.2**	-1.8	-0.9
290ML ethanol produced from C molasses using co-generated energy							
Upstream	34,398	2,299	-47	-91	-7.53	-3.5	-1.7
Tailpipe	-311,313	-55,531	1,087	2,360	-1.03	-0.2	-0.1
Total	**-276,915**	**-53,232**	**1,040**	**2,269**	**-8.6**	-3.7	-1.8

Note: Numbers may not add due to rounding.

Table 67 also shows the impact on total health costs of assuming all of the extra ethanol is produced from C molasses using cogeneration. In this case, because most of the extra particle matter is emitted in relatively sparsely populated areas, the health cost impact is relatively smaller. The total health cost saving of all the extra ethanol being produced from C molasses using cogeneration would result in total health cost savings of $A3.7 million in 2010, a saving of 1.8c/L.

13.1.2 Impact on Upstream Emissions of Using Gas rather than Coal

The upstream emissions rates applying to the production of ethanol used in the main scenario are based on existing ethanol production processes in place in Australia. At present, a principal source of energy used for raising steam is coal. Similarly, much of the electricity consumed is coal fired. However, gas-fired steam generation and gas-fired electricity are alternative sources of energy that could be used in ethanol production. Gas-fired generation is identified as the principal energy source by proponents of new grain-based ethanol production facilities.

Natural gas generally produces less greenhouse emissions, and lower particle emissions, than coal, per unit of energy. Consequently, it is expected that the use of gas instead of coal in the production of ethanol would produce less greenhouse and PM emissions than estimated under the main scenario. Because the ethanol production from C molasses using cogeneration scenario (above) provides the best-case change in upstream emissions from additional ethanol production, the use of gas instead of coal fired energy sources would result in a change in emissions somewhere between that and the estimate from the main scenario.

13.1.3 Impact on Emissions of Use of BD20 and BD5 Biodiesel Blends

The sensitivity of the transport emissions results to the choice of biodiesel blend concentration was tested by considering total tailpipe emissions arising from the combustion of the additional 30ML of biodiesel as BD20 and BD5 blends in rigid trucks. For the analysis, it is assumed that BD20 and BD5 fuels are blended with the prevailing standard diesel fuel, which is assumed to be ULSD from 2005 and XLSD from 2009.

It is also assumed that blending biodiesel and diesel fuels does not require any extra energy input, either directly in the blending process or indirectly (say through increased transport to blend sites, for example). Hence, the upstream emissions would be unaffected by the choice of biodiesel-diesel blend composition. Indeed, Beer *et al.* (2001, p. 162) suggest that upstream emissions from such blends can be calculated by using a linear relationship with diesel and the corresponding biodiesel emissions, weighted by their proportions. Only tailpipe emissions will be affected by the choice of biodiesel blend composition.

Tailpipe emission rates for biodiesel blends BD20 and BD5 are lower than for LSD, ULSD and XLSD for greenhouse gases and all pollutants, except NO_x. (The relative emissions performance of BD20 and BD5 blends, in comparison to LSD, ULSD and XLSD are outlined in Chapter 10).

Table 68 shows the change in tailpipe, and total, emissions assuming that the 30 ML of additional biodiesel is consumed as part of a blend fuel: BD20 or BD5. The emissions test results imply that if the additional 30 ML of biodiesel were consumed as BD20, the reduction in greenhouse emissions and total CO, VOC and PM emissions would be larger than if the extra biodiesel is consumed as BD100. In particular, the saving in total PM emissions would be three times as large for BD20 (15.1 tonnes) as for BD100 (5.8 tonnes). Total NO_x emissions, however, are higher if the additional biodiesel is consumed as BD20 than is the case if it is consumed as BD100.

If, instead, the additional 30ML of biodiesel were consumed as BD5 the change in pollutant emissions is larger, than is the case for BD100 and BD20, for all pollutants except PM. For BD5, the saving in tailpipe CO and VOC emissions in 2010 are 1,659 tonnes and 209 tonnes, respectively, while NO_x tailpipe emissions would be even higher, 1,526 tonnes in 2010. The change in total PM emissions (3.94 tonnes in 2010), however, is less than if the biodiesel were consumed as BD100 or BD20. The reduction in greenhouse gas emissions, however, is not as large as in the BD5 case, and NO_x emissions are significantly higher.

Table 68. Impact of BD20 and BD5 on GHG and pollutant emissions in 2010

	Greenhouse	**Pollutants**				**Health costs**	
	CO_2-e	**CO**	**NO_x**	**VOCs**	**PM**	Total	Average
	(tonnes)		**(tonnes)**			($m)	(c/L)
Upstream							
All blends	-8,934	-16	-87	-35	-1.27	-0.5	-1.7
Tailpipe							
BD100	-75,018	-208	173	-19	-5.85	-1.0	-3.5
BD20	-86,292	-518	423.4	-97	-15.08	-2.7	-9.1
BD5	-69,505	-1,659	1,526	-208.5	-2.67	0.8	2.5
Total							
BD100	-83,952	-224	86	-53	-7.12	-1.5	-5.2
BD20	-95,226	-534	336	-132	-16.35	-3.2	-10.8
BD5	-78,439	-1,675	1,439	-243	-3.94	0.3	0.9

Note: Numbers may not add due to rounding.

Table 68 also shows the impact on health costs of the alternative biodiesel blend options. The results imply that when used as a BD20 blend the 30 ML of additional biodiesel supply would result in total health cost savings of $A3.2m in 2010 (10.8c/L), compared with $A1.5m (5.2c/L) when the additional 30 ML is consumed as pure biodiesel. In contrast, use of the additional 30 ML of biodiesel in BD5 blend fuel would result in a total health cost increase of $A0.3m (0.9c/L) in 2010, as the cost of increased NO_x emissions more than offset the benefits of reduction in PM output.

These results should be treated with caution, as there have been few studies that have investigated the emissions performance of vehicles using biodiesel blended with ULSD and XLSD. Further research is required to draw firm conclusions as to relative emissions performance, and hence health cost impact, of pure biodiesel and biodiesel blends with ULSD and XLSD.

13.1.4 Impact on the Location of Emissions of Different Consumption Assumptions

The main scenario assumed that most of the ethanol and all of the extra biodiesel is consumed in metropolitan areas. The sensitivity of vehicle (tailpipe) emissions to these assumptions is tested with two cases: one where all the extra ethanol is assumed to be used in metropolitan areas and the other where half of the additional ethanol and biodiesel supply is consumed outside metropolitan areas.

Because the relative emissions rates between biofuels and conventional fuels are assumed to be invariant to the urban-rural drive cycle, the assumptions about the location of ethanol and biodiesel consumption will not have any effect on the total quantity of emissions. However, the location of consumption does affect total health costs.

Table 69 shows the estimated tailpipe emissions, for the extra ethanol and biodiesel supply, by area of use for the main scenario and the two alternative cases.

Table 69. Change in greenhouse and pollutant tailpipe emissions resulting from consumption of 350 ML of biofuels in 2010, by location (tonnes)

	Urban areas				**Rural areas**				***All areas***			
	CO	**NO_x**	**VOCs**	**PM**	**CO**	**NO_x**	**VOCs**	**PM**	**CO**	**NO_x**	**VOCs**	**PM**
Main case												
Ethanol	-49,978	977.9	2,124	-0.93	-5,553	108.7	235.9	-0.10	-55,531	1,086.6	2,360	-1.03
Biodiesel	-208	173.3	-19	-5.85	0	0	0	0	-208	173.3	-19	-5.85
Total	-50,185	1,151.2	2,105	-6.78	-5,553	108.7	235.9	-0.10	-55,738	1,260	2,341	-6.88
All additional biofuel supply consumed in metropolitan centres												
Ethanol	-55,531	1087	2,360	-1.03	0	0	0	0	-55,531	1,087	2,360	-1.03
Biodiesel	-208	173	-19	-5.85	0	0	0	0	-208	173	-19	-5.85
Total	-55,738	1260	2,341	-6.88	0	0	0	0	-55,738	1,260	2,341	-6.88
Half of additional biofuel supply consumed outside metropolitan centres												
Ethanol	-27,765	543	1,180	-0.52	-27,765	543	1,180	-0.52	-55,531	1,087	2,360	-1.03
Biodiesel	-104	87	-9	-2.93	-104	87	-9	-2.93	-208	173	-19	-5.85
Total	-27,869	630	1,171	-3.44	-27,869	630	1,171	-3.44	-55,738	1,260	2,341	-6.88

The case where all of the additional ethanol is used in major metropolitan areas produces slightly higher total health costs savings in 2010, $A1.82m as against $A1.79m. (These savings are not apparent in Table 70 because they are below the reported level of significance). The impact on total health costs of half the additional ethanol and biodiesel being consumed outside major metropolitan areas would result in slightly lower savings in total health costs, $A2.9m (1.2c/L) versus $A3.3m

(1.4c/L) in 2010. The reduction in avoided health costs is mainly due to a smaller reduction in urban PM emissions.

Table 70. Sensitivity analysis results – Area of use impact on health costs resulting from 350 ML biofuels consumption in 2010

	Health costs	
	Total ($m)	**Average (c/L)**
Main case		
Ethanol	-1.8	-0.9
Biodiesel	-1.5	-5.2
Total	-3.3	-1.4
All additional biofuel supply consumed in metropolitan centres		
Ethanol	-1.8	-0.9
Biodiesel	-1.5	-5.2
Total	-3.3	-1.4
Half of additional biofuel supply consumed outside metropolitan centres		
Ethanol	-1.8	-0.9
Biodiesel	-1.1	-3.8
Total	-2.9	-1.2

13.1.5 Overview of health costs – Scenario analysis

Table 71 provides a comparison of the impact on total health costs of the scenarios presented in this section.

The sourcing of all additional ethanol from C molasses, produced using co-generated energy, would result in the largest health cost savings, $A5.2m in 2010 (2.2c/L). The reduction in total greenhouse emissions is also largest for this case.

The tailpipe emission rates for BD20 imply much larger reductions in pollutant emissions from the use of an additional 30 ML of biodiesel blended with diesel than for the case where the extra biodiesel is use as 100 per cent biodiesel (BD100). Total health cost savings could be as high as $A5.0m in 2010 (an average saving of 2.1c/L of total additional biofuel supply) if all the extra biodiesel were consumed as BD20. However, the emission rates for BD5 imply slightly smaller health cost savings, around $A1.5m in 2010 (an average saving of 0.6c/L of additional biofuels supply). The blending of biodiesel has only a minor impact on total greenhouse emissions.

Altering the assumptions about where the extra biofuel supply is consumed has only a minor impact on estimated total health costs and no impact on total greenhouse emissions.

Table 71. Impact on total health costs of alternative sensitivity scenarios

	Health costs	
	Total ($m)	**Average (c/L)**
Main scenario	-3.3	-1.4
All additional ethanol sourced from C molasses using co-generated energy sources		
C molasses	-5.2	-2.2
Alternative biodiesel blend use		
BD20	-5.0	-2.1
BD5	-1.5	-0.6
Area of use		
All metropolitan	-3.3	-1.4
Half metropolitan	-2.9	-1.2

Note: Figures may not add to total due to rounding.

13.2 Probabilistic Uncertainty

The international comparison undertaken in this study provided information on the reduction of greenhouse gases and air pollutants and data for the evaluation of uncertainty. Insufficient Australian testing data for all biofuels led CSIRO to use previous studies in translating reductions in emissions between different groups of vehicles. The assumption was that the vehicles behave similarly in terms of emissions production and their response to different operating conditions. However, because of different vehicle types, sizes, and ages, driving cycles and units of measurement of various emissions, relative reductions and/or increases in emissions were used when estimating the variability of present emission rates.

This section presents a summary of the results of an examination of probabilistic uncertainty (variation) associated with the estimated change in emissions between the reference case and the 350 ML case level of biofuel supply and use in 2010. Further discussion of these results is presented in Appendix X.

13.2.1 Measured Variation in Emission Rates

The best estimates were determined from the following sources:

- APACE (1998) for ethanol;
- Graboski *et al.* (1999), Sharp (1998), and US EPA (2002) for biodiesel.

Test results indicated both high variances (within vehicle and between vehicles) for the same test condition, as well as different tests and fuels. Hence, large amounts of data are required to develop reliable emission rate models that can identify the changes due to alternative biofuels.

The results presented in Chapter 10 are point estimates and they indicate that ethanol E10 results in a 1.7% (wheat) to 5.1% (molasses cogeneration) reduction of exbodied greenhouse gas emissions per km compared to unleaded petrol (Table 29). In terms of criteria air pollutants, the results in Table 30 indicate that:

- CO emissions are reduced in a range from 20.9% (wheat) up to 26.2% (sorghum);
- changes in NMVOC emissions vary from -1.6% (sorghum) to 0.8% (wheat);
- NO_x emissions are generally higher by up to 6.2%;
- total PM emissions are increased by up to 39.1% (except molasses co-generation, for which PM emissions are reduced by 4.4%).

Analogously, the results in Table 31 indicate that replacing diesel with biodiesel BD100 in rigid trucks leads to overall reductions in exbodied GHG emissions per km of up to 23.3% (canola), 29.3% (tallow), and 89.5% (waste oil).

The use of BD100 results in reductions in CO emissions of up to 27.4% (canola), 36.8% (tallow), and 47% (waste oil), as shown in Table 34.

The difference in exbodied NO_x emissions for biodiesel and diesel are highly dependent on the feedstock for biodiesel and the base diesel. The comparison between BD100 and LSD shows similar emission rates on a per km basis for rigid trucks. Nevertheless, when the sulfur content diminishes to 10 ppm (i.e. XLSD), NO_x emissions from BD100 are between 15% (tallow and waste oil) and 30% (canola) larger than diesel.

BD100 presents certain benefits over diesel in terms of NMVOC emissions. When compared to LSD, BD100 provides reductions in NMVOC per km of between 32.2% (canola) and 49.8% (waste oil). These benefits are reduced by 7-10% when the sulfur content of diesel becomes less than 10 ppm. Finally, PM reductions of 32% (canola and tallow) to 39% (waste oil) provided by BD100 over LSD become reductions of 11% (canola) to 20% (waste oil) of BD100 over XLSD.

The considerable variation associated with the observed tailpipe emissions and the uncertainty associated with the upstream (pre-combustion) manufacturing processes are indicative that it is not possible to claim a statistically significant difference between ethanol E10 and unleaded petrol (ULP) or between biodiesel and diesel at the 95% confidence level without detailed data analysis.

CSIRO used the software @Risk (Palisade Corporation) to perform an uncertainty analysis and examine the probability distribution associated with the percentage difference between the biofuels and reference/base fuels. The uncertainty in the life cycle analysis (LCA) parts is summarised using triangular probability distributions. The testing data, as well as international experience, suggested that this distribution is a good approximation of uncertainties in the processes under study.

The results on uncertainty of emissions per km travelled are shown in Appendix X. In this section we present only the distribution of changes in the full fuel life cycle vehicle emissions (and health costs associated with them) from use of ethanol and pure biodiesel in 2010. The variation in emission rates is reflected in the estimated change in total vehicle emissions, and provides a measure of whether the increased use of biofuels results in a significant impact on total emissions. This analysis assumes a deterministic forecast of vehicle use till 2020.

The BTRE vehicle model treats separately the upstream and tailpipe emissions. The two modules use as input: emissions the lifecycle emission rates (upstream) and ratios of emission rates of the biofuel over the reference fuel (tailpipe emissions). This implies a statistical adjustment of variance of the ratio of emission factors in the tailpipe module.

Table 72 shows the expected value of change in emissions in 2010 and the 5 and 95 percentiles for the change in emissions as a result of the increase in ethanol and biodiesel use required to obtain 350 ML in 2010. The relevant findings are outlined below.

13.2.1.1 Ethanol

The greenhouse gas emissions from use of ethanol vary from -444 kt to 55 kt; this is driven by the opposite effects of E10 on the upstream (increase) and combustion (decrease) emissions; there is an 11% probability that the total greenhouse gas emissions will increase from the use of ethanol.

The E10 impact on tailpipe PM emissions is even more uncertain; the sparse testing data available for this study suggest that PM emissions resulting from combustion of E10 are similar to those from combustion of ULP. However, the vehicle emissions test results suggests that the tailpipe PM emissions from E10 can be higher or lower than the petrol PM emissions. In 2010, the use of E10 would result in an increase of PM emissions within the range 41 to 59 tonnes. Such a wide range of variation in PM would have a considerable impact on the range of variation in health costs.

The uncertainty is smaller in the case of CO emissions; these emissions will diminish between 49 and 62 kt in 2010.

Larger uncertainties are observed for the increased NO_x and NMVOC emissions from the use of E10.

13.2.1.2 Biodiesel

The assumed increased use of biodiesel (BD100) (30 ML in 2010) would reduce total exbodied greenhouse gas emissions by between 79 and 90 kt. Total CO emissions would be reduced by between with 200 and 247 tonnes in 2010 and exbodied NMVOC emissions would fall by between 43 and 64 tonnes.

For NO_x, upstream emissions decrease but tailpipe emissions increase as a result of the increased biodiesel use. The combined effect of the reduced upstream and increased tailpipe NO_x emissions results in an increase of between 45 and 127 tonnes of NO_x in 2010.

The increase in total biodiesel use would reduce the full life cycle PM emissions, the savings being between 4 and 10 tonnes; the main benefits appear from the combustion of fuels; although the introduction of XLSD will reduce significantly the PM emissions, there will still be PM reductions from using biodiesel.

Table 72. Uncertainty analysis - Change in greenhouse and criteria pollutant emissions resulting from consumption of 350 ML of Biofuels in 2010 (tonnes)

	Mean estimates					**5th percentile**					**95th percentile**				
	Green-house	**Pollutants**				**Green-house**	**Pollutants**				**Green-house**	**Pollutants**			
	CO_2-e	CO	NO_x	VOCs	PM	CO_2-e	CO	NO_x	VOCs	PM	CO_2-e	CO	NO_x	VOCs	PM
Ethanol															
Upstream	126,980	1,723	272	-44	51.22	121,575	1,577	185.4	-55.6	45.5	132,061	1,876	359.2	-32.4	56.8
Tailpipe	-311,313	-55,531	1,087	2,360	-1.03	-570,034	-61,276	43.8	1,786.8	-8.1	-69,795	-49,956	2,112	2,931.4	5.9
Total	-184,333	-53,807	1,359	2,316	50.20	-444,377	-59,489	309	1,733	41.3	55,834	-48,229	2,395	2,888.0	58.7
Biodiesel															
Upstream	-8,934	-16	-87	-35	-1.30	-9,875	-20.3	-102	-40.3	-1.5	-8,003	-11.9	-71.96	-29.3	-1.03
Tailpipe	-75,018	-208	173	-19	-5.85	-80,289	-230.1	135.1	-27.6	-9.2	-69,859	-185.4	211.2	-9.6	-2.47
Total	-83,952	-224	86	-53	-7.10	-89,321	-246.0	45.4	-65.3	-10.5	-78,645	-201.2	126.8	-43.2	-3.7
Biofuels															
Upstream	118,046	1,707	185	-79	49.92	112,712	1,556.2	99.4	-91.9	44.4	123,005	1,868	269.5	-65.3	55.5
Tailpipe	-386,331	-55,739	1,260	2,341	-6.88	-635,407	-61,478	213.5	1,768.3	-14.5	-147,327	-50,165	2,285	2,913.1	0.6
Total	-268,278	-54,031	1,445	2,263	43.10	-525,815	-59,745	396.8	1,737	33.2	-26,698	-48,763	2,496	2,882.1	52.1

13.2.1.3 Total biofuel results

Overall, the increased use of biofuels that would result from obtaining 350 ML of total biofuel supply in 2010, would result in overall reductions in greenhouse and CO emissions, and increases in the remaining air pollutant emissions, that are all significantly different from zero.

Figures 15 to 19 display the changes in the full fuel life cycle emissions of biofuels use in 2010. Figure 20 illustrates the distribution of the tailpipe PM emissions from use of biofuels in 2010.

The distribution of changes in GAG and PM emissions, vary considerably. The range of variation in total tailpipe PM emissions shown in Figure 20, which are estimated to vary between -15 to 1 tonnes in 2010, reflects the large uncertainty associated with the combustion of ethanol. The uncertainty results indicate there is a 7 per cent probability that total tailpipe PM emissions would increase in 2010 as a result of the assumed increase in biofuel supply.

All charts represent standard histograms that give a complete picture of all possible results. The charts show the expected value (mean), the range (min to max), and the relative likelihood of occurrence for each possible outcome.

The two vertical lines are called ‘sliding delimiters’ and are used for setting and/or comparing probabilities for results. By default they are set at 5% and 95%; in this case they indicate a 0.05 probability that the changes in vehicle emissions between the reference and target cases are lower than the value printed on the left hand side, and a 0.05 probability that the changes are higher than the printed value at the right delimiter.

By dragging the delimiters, one can calculate target probabilities. This is particularly useful for graphically displaying answers to questions such as: “What is the probability of a negative result (increase in greenhouse gas or air pollutant emissions) occurring in 2010?” Figure 20, for example, shows that the probability of an increase in tailpipe PM emissions is 0.07.

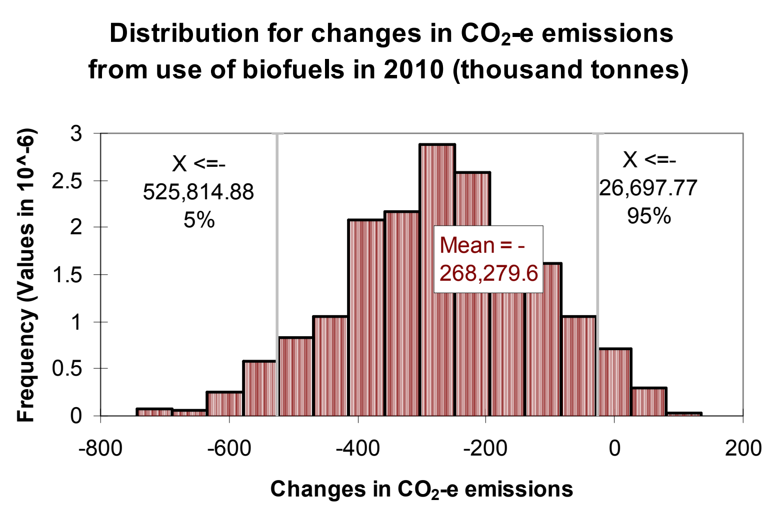

Figure 15. Distribution of changes in GHG emissions in 2010 from use of biofuels: Difference between reference case and 350 ML biofuels consumption case

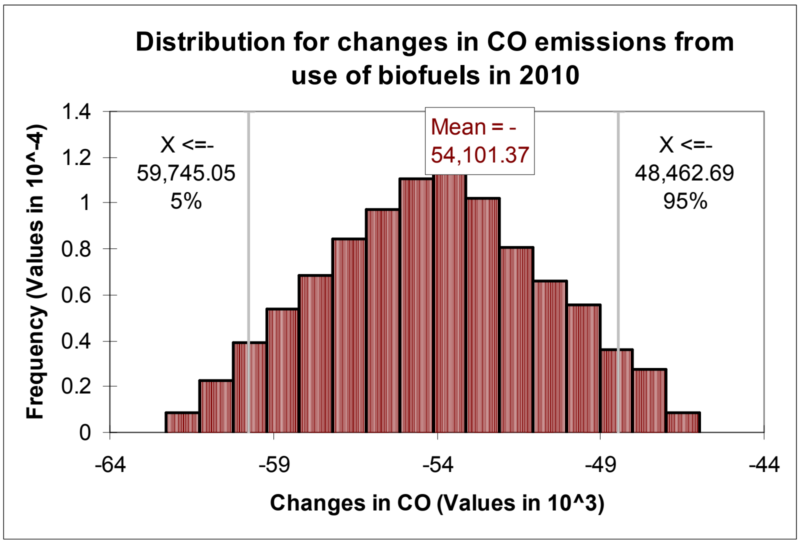

Figure 16. **Distribution of changes in CO emissions in 2010 from use of biofuels: Difference between reference case and 350 ML biofuels consumption case**

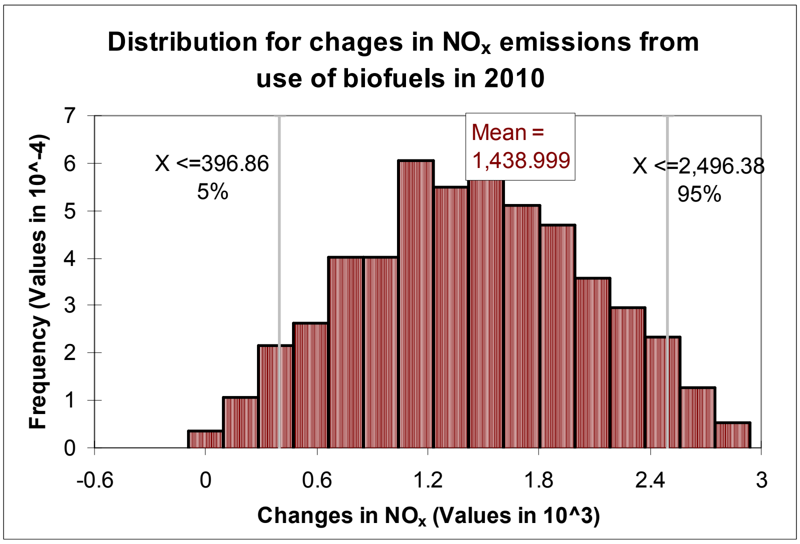

Figure 17. **Distribution of changes in NO_x emissions in 2010 from use of biofuels: Difference between reference case and 350 ML biofuels consumption case**

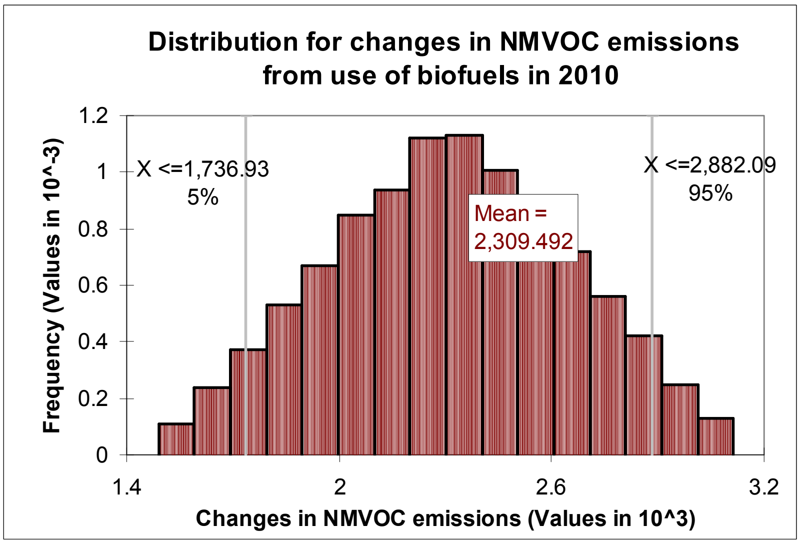

Figure 18. **Distribution of changes in NMVOC emissions in 2010 from use of biofuels: Difference between reference case and 350 ML biofuels consumption case**

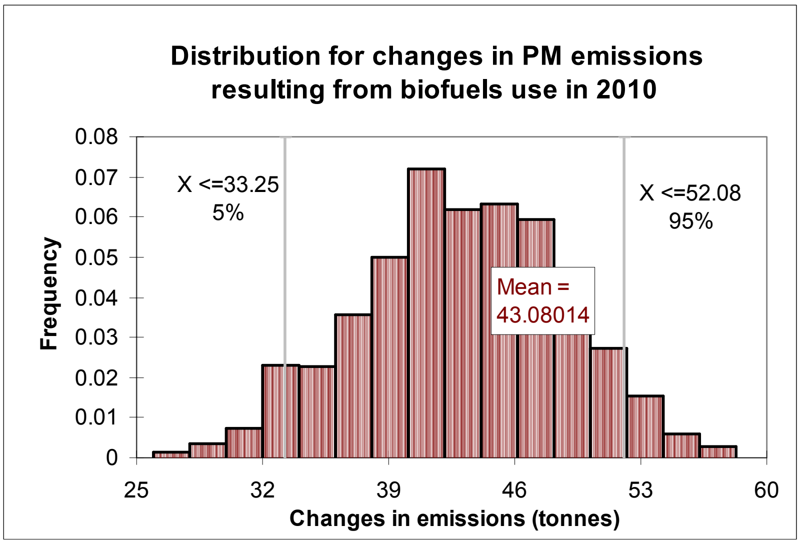

Figure 19. **Distribution of changes in PM emissions in 2010 from use of biofuels: Difference between reference case and 350 ML biofuels consumption case**

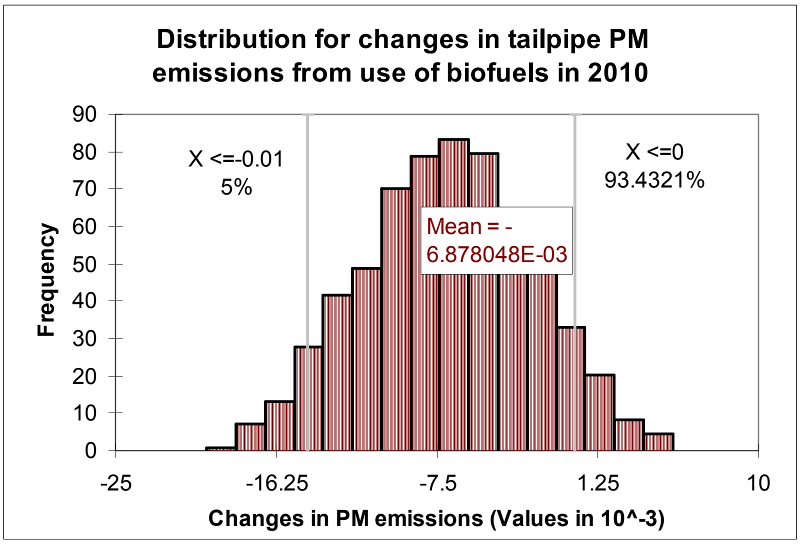

Figure 20. **Distribution of changes in tailpipe PM emissions in 2010 from use of biofuels**

13.2.2 Health cost uncertainty

The variation in total vehicle emissions implies considerable variation in the estimated health costs. Figures 21 to 23 display the variation in health costs resulting from use of E10, pure biodiesel, and their combined effect in 2010. The variations in health costs relate to the uncertainty associated with the upstream and tailpipe emissions and the assumed uncertainty associated with the monetary health costs.

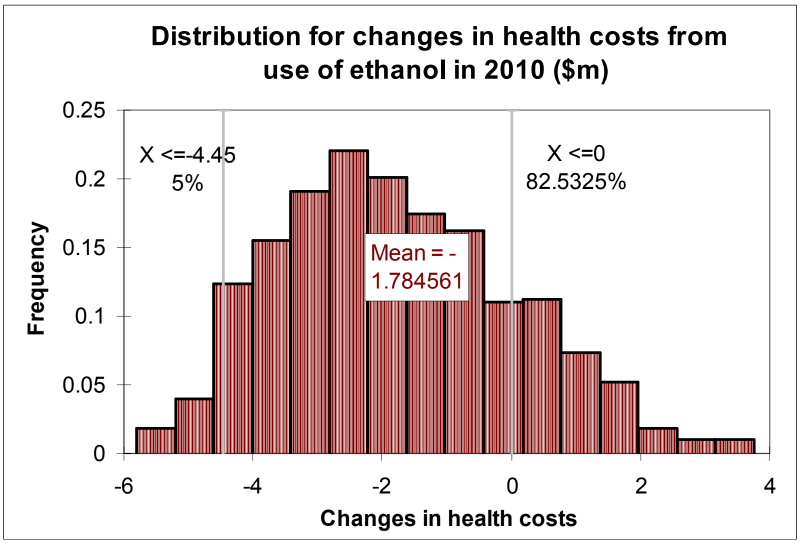

Figure 21. **Distribution of changes in health costs from use of ethanol in 2010: Difference between reference case and 350 ML biofuels consumption case**

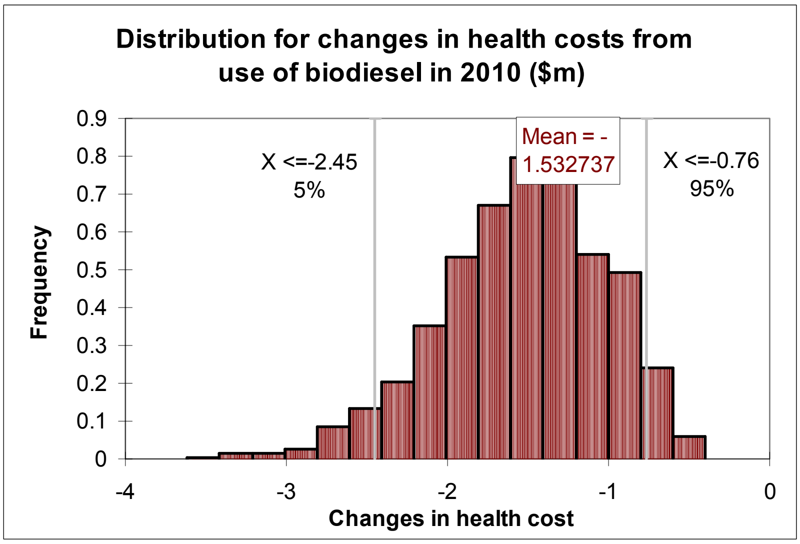

Figure 22. **Distribution of changes in total health costs from use of biodiesel in 2010: Difference between reference case and 350 ML biofuels consumption case**

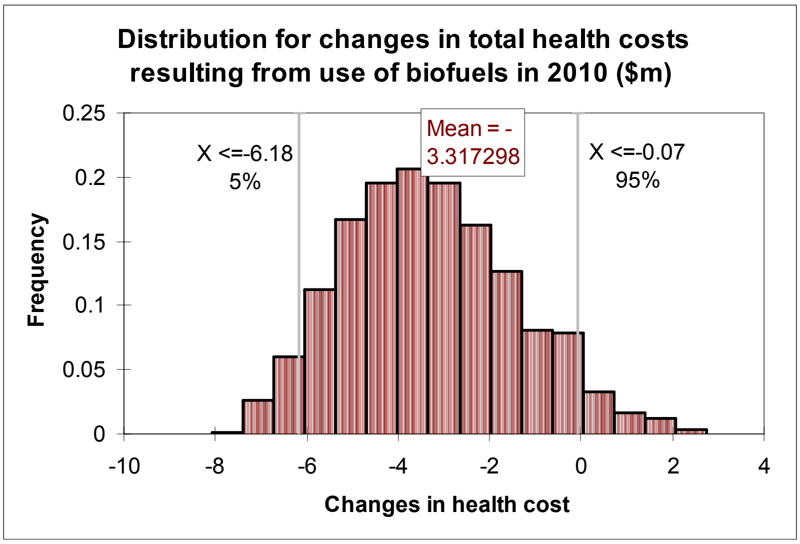

Figure 23. **Distribution of changes in total health costs from use of biofuels in 2010: Difference between reference case and 350 ML biofuels consumption case**

The charts as well as results presented in Table 73 show that the avoided health costs have a large variability, arising from two sources already mentioned: emission rates and monetary health costs.

Although the mean values represent savings in avoided health costs (benefits), it should not be overlooked that there is some probability of having increased health costs due to the increase in total biofuel use: 12% for emissions resulting from upstream processes, 10% for emissions from combustion.

In the most pessimistic situation, the increased biofuels use implies no change compared with the ‘business-as-usual’ case. In the optimistic situation, the increased biofuels use will lead to $6.2m savings in health costs, whereas the most likely savings are $3.3m. These values are strongly determined by the value assigned to the various criteria pollutants, as the regression sensitivity analysis in Section 13.3 will show.

Table 73. Uncertainty analysis - Change in health costs resulting from consumption of 350 ML of Biofuels in 2010 ($m)

	Avoided health costs ($m)			probability of increased health costs
	Mean estimate	5th percentile	95th percentile	
Ethanol				
Upstream	-1.584	-3.858	1.217	0.165
Tailpipe	-0.220	-1.673	1.317	0.407
Total	-1.81	-4.452	1.440	0.175
Biodiesel				
Upstream	-0.494	-0.664	-0.305	0
Tailpipe	-1.038	-1.919	-0.2961	0
Total	-1.532	-2.451	-0.761	0
Biofuels				
Upstream	-2.078	-4.328	0.680	0.115
Tailpipe	-1.260	-2.869	0.392	0.099
Total	-3.337	-6.177	-0.069	0.047

13.3 Sensitivity Analysis

The effects of changes in emission rates and health costs on model predictions have been quantified, accounting for the uncertainty associated with production, transport, distribution, combustion of fuels, and the uncertainty related to the avoided health costs.

The sensitivity analysis presented here identified the most significant inputs in the avoided health costs. The impacts of controlled/deterministic input variables have been examined by the sensitivity simulation that accounts for stochastic elements. The sensitivity analysis used probability distributions to describe the uncertain variables in the model: emission rates and unitary (monetary) avoided health costs[23].

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[23] We used both Latin Hypercube and Monte Carlo sampling.

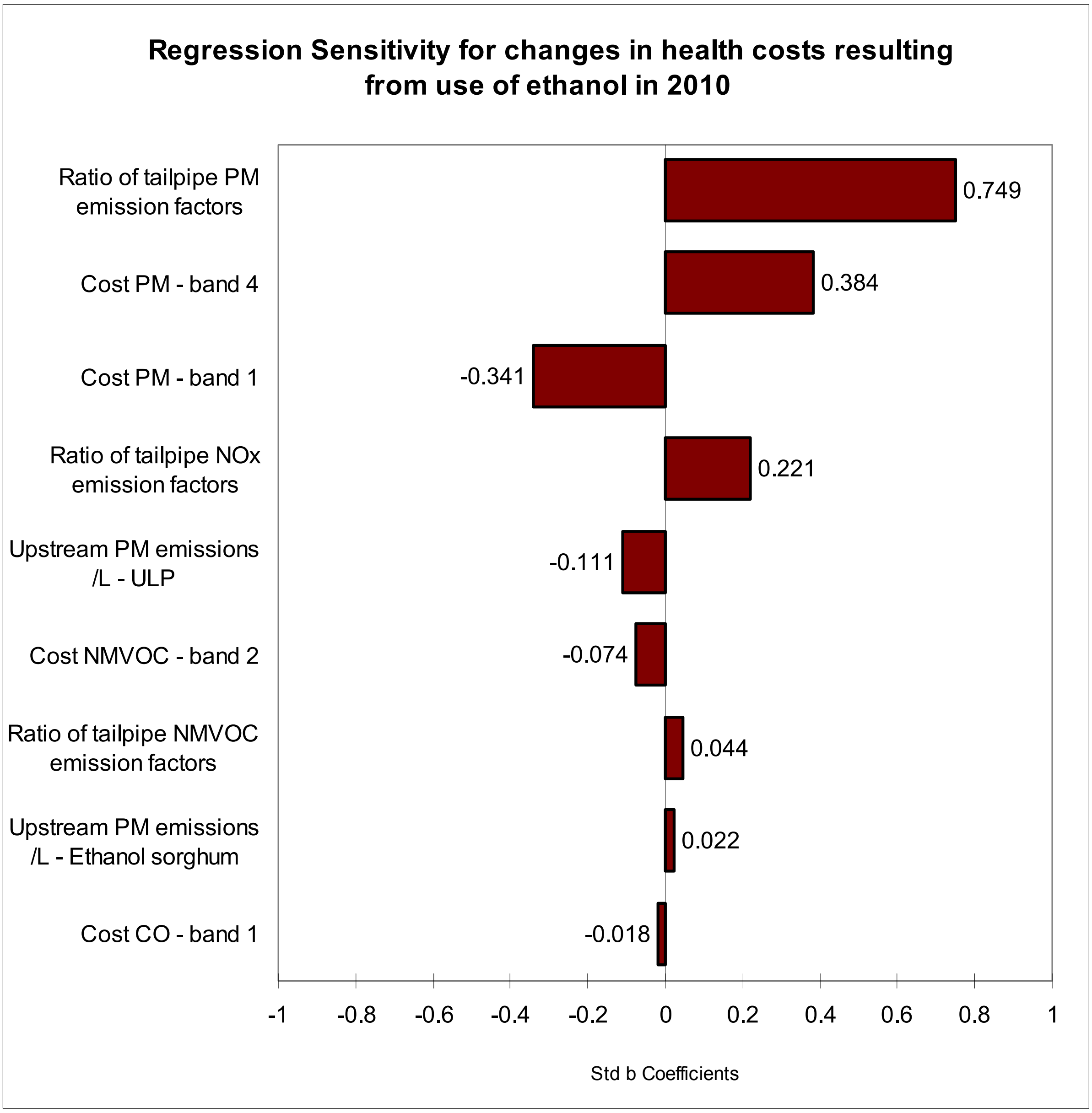

Figure 24. Regression sensitivity for health costs from use of ethanol in 2010: Difference between reference case and 350 ML biofuels consumption case

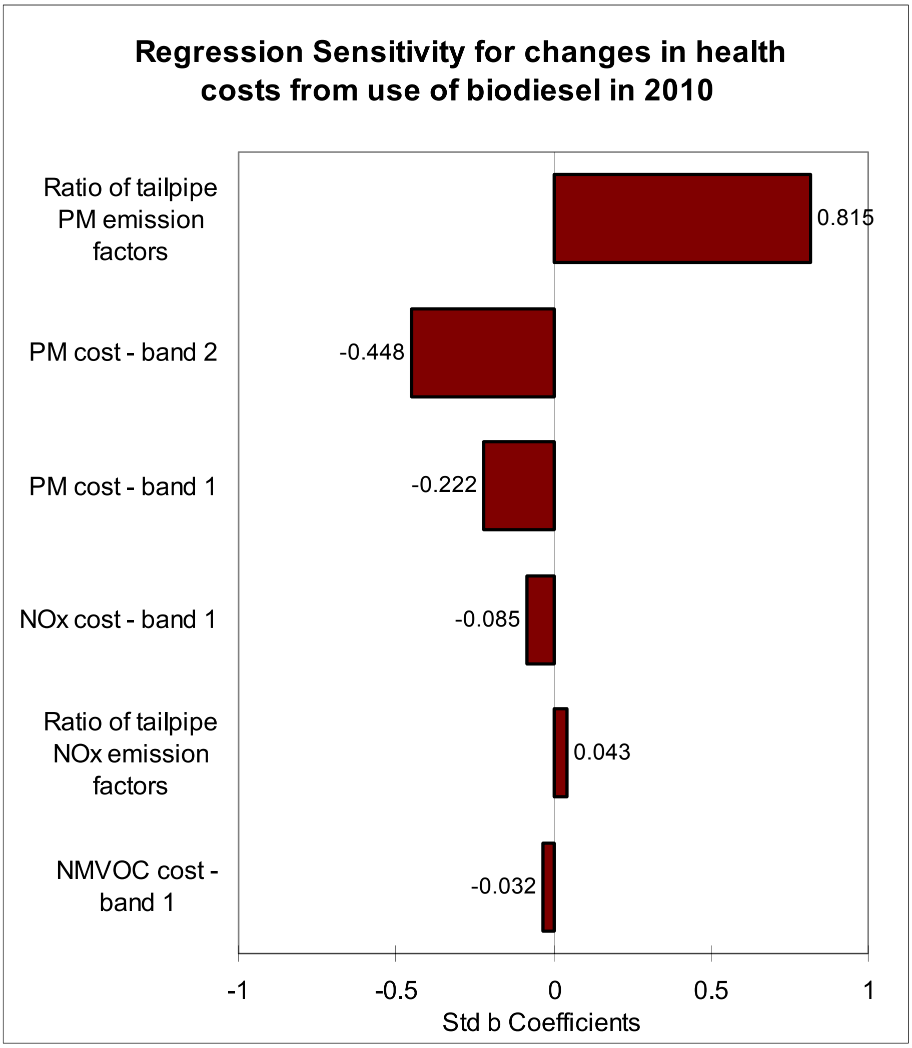

Figure 25. Regression sensitivity for health costs from use of biodiesel in 2010: Difference between reference case and 350 ML biofuels consumption case

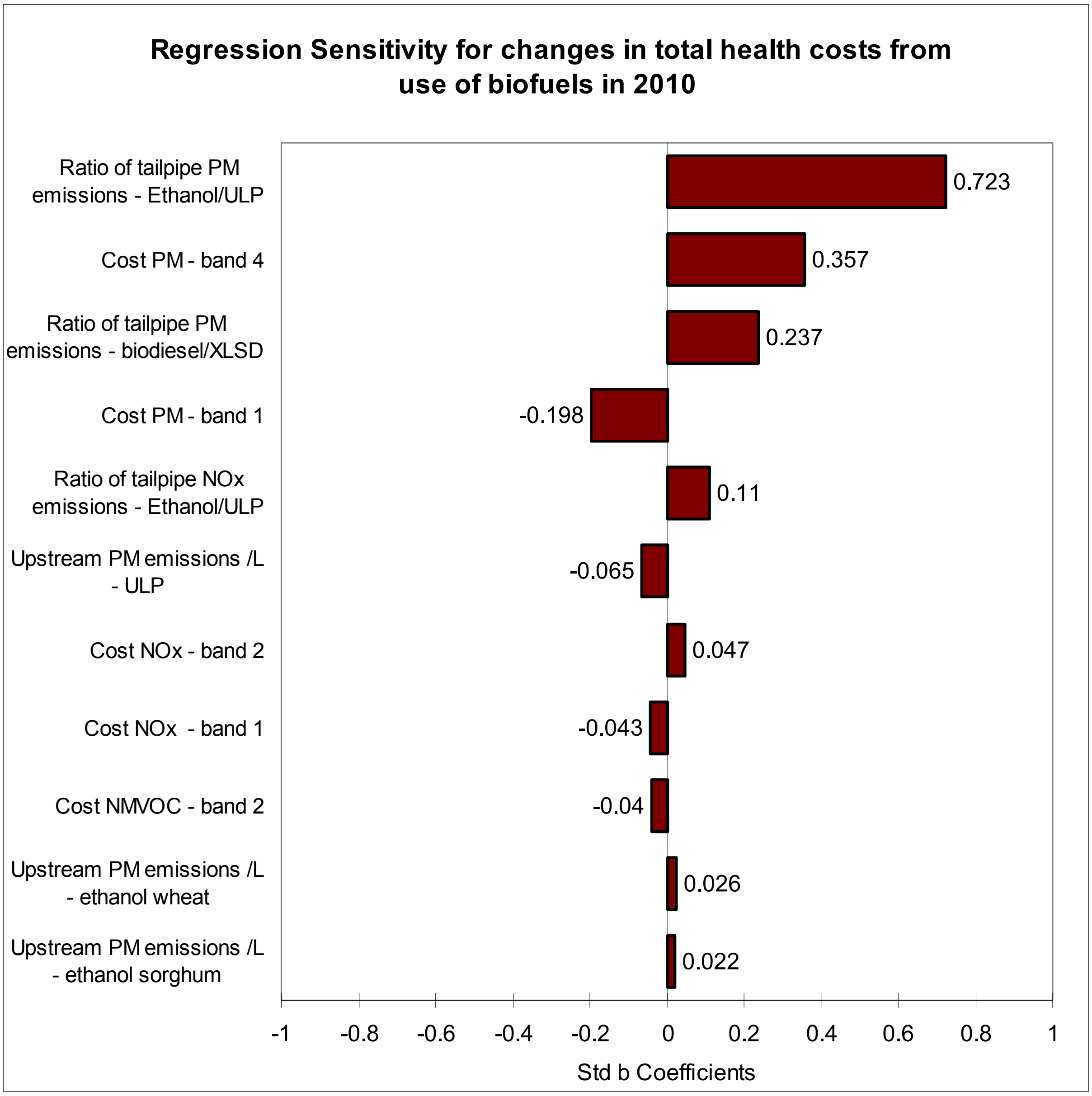

Figure 26. Regression sensitivity for health costs from use of biofuels in 2010: Difference between reference case and 350 ML biofuels consumption case

The analysis was carried out using multivariate regression analysis (the R^2 values resulting from the sensitivity analysis range between 0.96 and 0.998). The results are displayed as tornado type charts (Figures 24 to 26), with longer bars at the top representing the most important input variables that contribute to the variability of the results.

The calculated coefficients that are shown in Figures 24 to 26 represent standardised coefficients that measure the change in output (in standard deviations) when the input changes by one standard deviation. For example, one standard deviation increase in the ratio of PM emissions from combustion of E10 over ULP increases the changes of health costs from ethanol use in 2010 by 0.749 standard deviations (Figure 24). One standard deviation increase in the health costs associated with PM emissions from petrol diminishes the relative health costs of ethanol use in 2010 by 0.341 standard deviations.

Figure 24 shows that the most influential factors for the changes in health costs in 2010 due to use of ethanol are the relative change of PM emissions from ethanol compared to petrol, followed by the costs for PM and relative change in NO_x emissions. The implications are that:

- improving the fuel-vehicle technologies for use of ethanol will diminish the burden of associated health costs of pollutants (PM and NO_x);
- adopting a lower value for health costs of PM emissions in urban areas will diminish the advantage of ethanol over petrol; similarly, a higher value for PM emissions in rural areas (where the production of ethanol is assumed to take place) will reduce the benefits of ethanol over petrol.

The first implication is most relevant: due to the lack of PM tailpipe data from Australia or data that is representative of Australian conditions, we assumed PM reductions from the use of ethanol to be 1% (average). If the reduction were to be 10%, the total PM emission savings from ethanol would vary between 23 and 108 tonnes, and the total impact on PM emissions from biofuels use in 2010 would be in the range -115 to +15.4 tonnes, with associated health cost savings up to \$31m (with an average of \$18.4m).

The PM emissions for passenger vehicles are in general low, therefore the concern in relation to pollutants arises from poorly maintained vehicles and heavy-duty vehicles. More testing is needed on blends with ethanol (10%) versus petrol, covering a range of vehicle types and group ages, especially for PM and NO_x measurements. The results also imply that the reduction of upstream PM emissions by using cogeneration will improve the environmental bottom line of ethanol.

In the case of biodiesel, the most influential factors that contribute to the changes in health costs are the ratio of tailpipe PM emissions of biodiesel and XLSD (0.815), and the PM costs (-0.448 and -0.222). Reducing the PM tailpipe emissions from combustion of biodiesel by using specific devices that control the exhaust emissions (particulate traps, catalysed exhaust filters) and following a constant maintenance program will decrease considerably the health costs of using biodiesel relative to XLSD.

The above conclusions hold also for total changes in health costs, shown in Figure 26. Benefits in avoided health costs can be obtained by reducing the PM exhaust emissions of biofuels with respect to the reference fuels (0.723 for E10 versus ULP and 0.237 for biodiesel versus XLSD) and reducing the upstream PM emissions from production of ethanol (0.022 to 0.026). Reductions in NO_x exhaust emissions of ethanol would also improve the health costs savings (0.11).

The dominance of ethanol impacts is mainly explained by the larger expected use of ethanol. Consequently there are higher emissions associated with it, in the 350 ML scenario.

13.4 Conclusions from the uncertainty and sensitivity analyses

The uncertainty and sensitivity analyses show large variability in the full life cycle emissions of all fuels. These uncertainties are mainly associated with the combustion and production phases of the life cycle (Appendix X) and they impact on the air quality.

Greenhouse gas emissions and CO emissions are reduced with the use of biofuels, but the increase in NMVOCs, NO_x and PM emissions offsets some of the overall benefit, resulting in average savings of avoided health costs of \$3.3m. The variability of the input elements shows that the change in health costs ranges between savings of \$0.07 and \$6.2m.

The variability of health costs is mainly influenced by the relative changes of PM exhaust emissions of biofuels compared to the reference fuels, and the unit health costs associated with the criteria air pollutants.

14 REGIONAL IMPACTS

14.1 Introduction

The aim of this chapter is to assess the net regional effects of replacing fossil fuels with biofuels in the Australian transport mix, as required in the Terms of Reference for this study.

For the purposes of this research, the analysis of 'regional' impacts refers to impacts on non-metropolitan areas of Australia: that is, areas outside the main capital cities.

The terms of reference and relatively short timeframes for this study mean that a review of existing literature from both Australia and overseas on the regional impacts of biofuels forms the main basis for the assessments made in this chapter. It is important to note that most of the existing literature on regional impacts relates only to ethanol. The study team is not aware of any information on the regional impacts of biodiesel plants being available.

The regional impacts of biofuels result from the construction and ongoing operation of the production plants rather than the products themselves. Impacts on economic activity typically include levels of output, value-added, efficiency, employment and income. Social impacts can be broad ranging and include, for example, increased community confidence, social cohesion and social capital. Economic and social impacts are often intimately interrelated with greater community confidence leading to demonstrated community leadership, which attracts new business opportunities, economic activity and employment.

The regional benefit claims associated with biofuels are heavily focused on employment impacts as these are often the most visible and easily understood impact. As a result, the discussion in this chapter will necessarily be focussed on employment, recognising, however, that there is a complex and broad set of potential impacts. Employment is also an impact that can be reasonably easily quantified (as are many of the economic impacts) unlike many other social benefits often put forward. There are also a number of other benefits attributed to biofuel production. For example, arguments relating to greater diversity in industry structure and agricultural crops giving communities a greater capacity to adapt to structural changes and price fluctuations.

Impacts refer to both benefits and costs. Most of the literature on biofuels concentrates on the benefits of such activity; however, there are also a number of costs and risks that some stakeholders have identified. Importantly, one of the key economic principles relevant to this discussion is the concept of national economy-wide impacts compared to regional impacts. For example, the benefits to one region can be achieved at the expense of those in another. A boost in employment in one region due to the construction and operation of a biofuel plant may lead to costs through reduced employment in another region. The net national effect could be either positive or negative. From a Commonwealth perspective, net national impacts must be considered. Other costs include higher input prices leading to reduced output and employment in other industries that may compete with biofuels for common inputs. The national economic impacts are examined in Chapter 15.

The following sections examine the existing Australian and overseas evidence on the regional benefits and costs of government support for biofuel production. This is followed by some broad estimates of the government expenditure per biofuel job created in Australia and an examination of potential locations. The chapter concludes by making some observations regarding the appropriateness of government support for biofuels based on regional development grounds.

14.2 Potential Regional Benefits

14.2.1 The Centre for Agricultural and Regional Economics (CARE) 2001 Study

This study is based on a particular form of biofuel production—ethanol—using sugarcane/molasses as the feedstock. The underlying motivation was a desire to develop greater stability in an industry buffeted by international sugar price volatility through product diversity and value adding. Based on this study, the *Biofuels for Cleaner Transport* policy identified expectations of at least five new ethanol plants being established, resulting in around 2,300 construction jobs and 1,100 permanent additional jobs, mostly in rural areas. This study is often quoted as the basis of general statements about expected regional benefits of additional biofuels production.

The CARE study was based on estimating the regional impact of the existing Sarina ethanol plant in the Mackay area of northern Queensland as an indication of the likely impact of the construction and operation of an ethanol plant. The plant has a capacity of approximately 60 ML of ethanol using sugarcane/molasses as its feedstock. The analysis is based on the plant's integrated operations of both ethanol and fertiliser.

The key findings of the CARE study were:

- that the plant created 36 direct permanent jobs and 222 flow-on (indirect) jobs (258 total jobs) for the Queensland economy;
- of this, the local impact was estimated to be 36 direct permanent jobs and 180 flow-on jobs (216 total jobs);
- one-off construction impacts of approximately 389 direct jobs, and 473 flow-on jobs (862 total jobs) for the Queensland economy as a whole;
- of this, the local impact was estimated to be 208 direct jobs and 256 flow-on jobs, (464 total jobs);
- for every permanent direct job created, around 6 flow on jobs are created, primarily in transport, chemicals and trade sectors;
- a plant of this size would add around $7.7m to household income in the region; and
- income tax revenue from this employment would be $1.915m and $0.9m in GST revenue.

14.2.1.1 Comments

There are a number of aspects which caution against accepting the CARE study estimates as a basis for generalised expectations about regional development benefits of biofuels production:

(a) The estimates are specifically related to the Sarina plant—its linkages with the Mackay and Queensland economies were inputs into the input-output model to produce specific multipliers rather than relying on generic industry multipliers. While this is good practice for the impact analysis of this particular case, it also means that the relevance or transferability of the Sarina experience to other proposed plant locations with different economic structures and linkages is doubtful.

(b) The CARE study used input-output analysis techniques to estimate impacts for the Mackay region and the Queensland economy using the 1996/97 Queensland input-output table. While the summary figures for employment and income effects are often quoted without qualification, in fact, the study itself states that the estimates for the Mackay local region are overestimated by around 20% as a result of the Queensland input-output table being used to project local impacts in the absence of a Mackay table. This means the local permanent impact would be around 29 direct jobs and 144 flow-on jobs.

(c) No analysis of the net national impact was undertaken. It is therefore not clear that the estimates relate to new jobs created rather than a transfer of jobs between industries or across locations. As a result, these local estimates are still likely to overstate the net national impact. For example, the existing ethanol plant at Sarina currently ships ethanol to the CSR facility at

Yarraville in Victoria for final processing. With the expansion and new facilities analysed in this study, this shipping would no longer be required. Given the limited nature of the CARE analysis to Queensland only, it is unlikely that any loss of employment in Victoria associated with the ceasing of this activity has been included. If this is the case, then the estimates are likely to overstate the employment benefit for the nation as a whole.

(d) The multiplier of six between direct jobs and flow-on employment is very high. Typically, a realistic multiplier impact is in the range of 1-2. The high multipliers are claimed to reflect the use of locally produced inputs and the low level of employment relative to output.

(e) The Sarina plant used as the basis of the estimates for general regional impact is capital intensive and results in a very high level of gross output per employee ($909,472 compared to the Queensland average of $122,000). The flow-on output is more labour intensive resulting in a much lower output per employee. The main flow-on impact is found to be in the transport (road), chemicals, wholesale and retail trade and other services sectors. This reflects the heavy use of roads for delivery of molasses and the spreading of dunder on cane fields. Dunder is specific to sugar feedstock and therefore flow-on impacts associated with it are not necessarily relevant for other ethanol plants based on different feedstocks.

(f) Construction is more labour intensive (lower output to employment ratio) therefore a smaller multiplier results—for every person employed in constructing the plant, there are a further 1.2 people employed in support activities. The construction of a new plant is estimated to cost $55m and the analysis assumes that 41% of construction is locally sourced, 37% is from within Queensland and 22% is imported. The ability of different regions to supply inputs for biofuel plants will vary considerably across locations depending on the size and economic base of each area. Consequently, the applicability of these employment estimates to other locations is uncertain.

(g) The CARE analysis relates to a sugarcane-based ethanol and fertiliser operation. While the estimates may be reasonable to apply to other sugarcane-based ethanol plants, their applicability to other feedstock sources, such as grains, is doubtful. At present, sugarcane-based ethanol contributes very little to biofuel use in transport and the analysis in Chapter 7 indicates that sugarcane is unlikely to be the major feedstock used in any future production. As a result, the CARE (2001) estimates may not be relevant to the most likely and most viable sources of biofuels production in Australia.

14.2.2 The CARE 1996 Study

CARE also investigated the potential regional impact of ethanol production in regional NSW. Their 1996 report includes three case studies of ethanol production based on various waste feedstocks. The results are summarised in Table 74.

Table 74. CARE 1996 study results

	Richmond-Tweed	Gwydir	Illawarra, Nowra
Feedstock	forest residues	cereal residues	wheat starch
Plant capacity (million litres)	50	50	80
Permanent direct jobs	34	34	6
Permanent flow-on jobs	288	125	357
Multiplier	8 (approx.)	4 (approx.)	60 (approx.)
Major flow-on industries	harvesting, hauling of materials	transport, trade	flour products processing
Construction -direct jobs	49	58	68
Construction -flow-on jobs	63	64	87

Source: CARE 1996.

On the basis of these estimates in this 1996 report CARE concluded that ‘any region where there is ready availability of residues could establish an ethanol plant and gain economic benefits. Potentially, some 30 plants could be established in regional areas of NSW’ (CARE, 1996, p. 27).

14.2.2.1 Comments

(a) The underlying assumption in all three cases that these waste feedstock sources have no economic value is critical to the study's results. The variation in impacts is found to be due to differences in plant size, source and cost of feedstock, and the economic structure of the regional economy (e.g. the capacity to source inputs from within the region). The estimates are stated to be conservative and relate only to the local region. The lack of national analysis of the net regional impact means that the criticisms made earlier are also relevant here. As a result, these local estimates are likely to overstate the net national impact.

(b) The number of direct jobs created is reasonably consistent with the 2001 CARE study. However, the extent of flow-on employment varies considerably reflecting a number of factors, including the difficulties inherent in forecasting such impacts. Flow-on employment is less in Gywdir than in the Richmond-Tweed due to the smaller size and less diverse nature of the Gywdir economy. In Nowra, the estimates relate to the expansion of an existing plant and the close integration with existing facilities and more highly automated plant nature means small direct employment effects. The flow-on impacts, however, are estimated to be extremely high with most flow-on employment resulting from an expected expansion of the starch/gluten mill. Even with the expansion of the flour mill, a multiplier of 60 appears overly optimistic. The employment generated during the construction phase is generally lower in these case studies than that estimated for the Sarina plant. Given the similar size of the proposed plants, this is likely a result of less construction inputs being sourced locally.

(c) The multipliers used in each case are reasonably high. According to the study, this is due to the low ratio of employment to output in the plants. Like the 2001 analysis, this report also assumes that these employment impacts are all new or additional activity ignoring any transfers to or from other regions of Australia. The net national impact remains uncertain.

14.2.3 Other Studies—Local

14.2.3.1 Ernst and Young (2002)

The Manildra Group and CSR Distilleries commissioned this report from Ernst and Young, which identified 36 full time employees working in each new ethanol plant. The combined direct public benefits are estimated at $49.4m (includes personal and company tax and reduced unemployment benefits). This research also found that the average annual net cost to Treasury over a ten-year grain ethanol plant production cycle is 20c/L in revenue forgone (not the total excise of 38.14c/L). The estimate for C molasses ethanol production was 23c/L in revenue forgone. As a result, the cost to Treasury of supporting the 350 ML biofuel target is argued to be in the order of $70 to $80.5m per annum (not $133.5m per annum, the gross revenue forgone based on the fuel excise of 38.14c/L).

The detailed modelling undertaken to support the above conclusions could not be reviewed. There are, however, a number of potential limitations to this analysis. If it is the case that the modelling has assumed there are no factor constraints in the economy, such that there is an unlimited and unconstrained supply of labour and capital, then the results are likely to understate impacts on the capital and labour markets. For example, it would be false to assume that any additional employment in the ethanol industry would lead to an equivalent reduction in general unemployment. New employment of labour and capital (in the ethanol industry) will have the effect of drawing labour and capital from elsewhere in the economy.

Local Proponents' Information

A range of information was provided by biofuel proponents regarding expected regional impacts. This information was provided on a commercial-in-confidence basis and therefore only a brief description of these mostly unquantified regional benefits follows:

- revitalisation of stressed regional and rural communities;
- increased investment and jobs in rural and regional Australia;

- more stable regional employment;
- economic revival in rural and regional Australia;
- increased value added to agricultural crops;
- product/crop diversification for farmers, making farmers less exposed to global commodity price fluctuations and other shocks;
- more efficient use of agricultural and forestry residues;
- alternative and stable income for growers (through fixed supply agreements that would provide a guaranteed market and price for a set volume of produce);
- more diverse regional industry base (through value-adding to primary production) providing more stable economic activity and greater resilience to economic shocks;
- establishment of an industry value chain including production and harvesting of feedstock, transport, processing, ancillary industries, exports, storage, distribution, maintenance etc;
- significant stimulus to the agricultural sector, leading to significant social benefits, increased government tax receipts etc;
- regional infrastructure enhancement and maintenance (e.g. improved local roads, extension of natural gas networks, revitalised regional rail services etc); and
- total government revenue from the ethanol industry estimated at $173m (including $59m in regional development jobs and tax benefits).

14.2.4 Other Studies—International

Various pieces of research from the United States on the regional development benefits of biofuels were also identified (although the detailed analysis and source behind these estimates could often not be located):

- experience in the US indicates that most of the income (80%) would be spent within a 100-200 kilometre radius of an ethanol plant, resulting in measurably stronger local economies and higher standards of infrastructure maintenance; and
- production of over 7 billion litres of ethanol per year in the US has created over 192,000 jobs in rural areas (US Renewable Fuels Association 2003);
- an average 40 million gallon per year ethanol plant will have the following positive economic impacts on a local community: a one-time boost of US$142m to the local economy during construction; expanded the local economic base of the community by US$110.2m each year through the direct spending of US$56m; 41 full-time jobs at the plant and a total of 694 jobs throughout the entire economy; increased household income for the community by US$19.6m annually; and provided an average 13.3% annual return on investment over ten years to a farmer who invests US$20,000 in an ethanol production facility (US Renewable Fuels Association, 2003).

Caution must be used when applying US results to biofuel potential in Australia. In the US, most biofuel is produced from corn (not molasses or grain as would be the case in Australia) and biofuels have been heavily assisted by governments (Yacobucci and Womach, 2003, p. 2). Many of the regional development benefits in the US have been the result of community-owned cooperative biofuel plants enabling more benefits to stay in the region. Cooperative models where farmers own or part own the downstream value adding have been a strategy for strengthening rural economies in the US (for example, in Minnesota) (Morris and Ahmed, 1993).

14.2.5 Conclusion

Overall, the regional benefits identified and quantified in the literature indicate the potential for significant gains. The estimates of direct employment impacts are reasonably consistent at around 30-40 jobs per plant (but did range from 6 to 50). The estimates of indirect flow-on employment impacts were also generally similar in the range of 180 to 200 jobs (but did range from 100 to 400). Construction employment varied widely from around 60 to 500. Estimates of future total employment resulting from a biofuels industry ranged from 3,000 to 10,000. Multipliers used to estimate indirect impacts varied considerably from 2 to 60 but were most commonly around the 5 to 6 range.

Despite the potential benefits illustrated by the literature, the estimates of benefits generally suffered from a number of important limitations casting doubt on the extent to which they should be relied upon in making policy decisions. Key limitations included:

- lack of analysis of net national impacts;
- the extent to which benefits are new employment versus displacing existing employment in other industries or locations;
- use of very high multipliers to estimate flow-on benefits; and
- lack of transferability of results given the location-specific nature of biofuel plants.

14.3 Potential Regional Costs

The available literature on potential negative impacts or costs associated with establishing a biofuels industry is generally more limited and tends to be predominantly qualitative. There are, however, a number of arguments put forward to counter the claims regarding the benefits of biofuels for regional development:

- subsidies distort markets and lead to inefficient outcomes, reduced employment and economic activity in other areas;
- a subsidised biofuel industry would compete directly and unfairly with other industries using the same inputs and those industries producing competing products (including by-products);
- subsidising biofuels can lead to the promotion of unsustainable development;
- diversification of crops and industries can mean that farmers and regional economies do not capture the benefits of high growth; and
- subsidising biofuels is an expensive, inefficient and indiscriminate approach to achieving regional development policy goals.

Underlying most of the arguments against subsidising the biofuels industry is the basic economic principle that subsidies can introduce inefficient distortions into the market and reduce the total value of economic output.

Government subsidies to support particular industries are problematic for many reasons. For example, subsidies can insulate the beneficiaries from the competitive pressures of the marketplace, reduce incentives to improve performance, leading to economic inefficiencies and the need for ongoing government support. By distorting the market, subsidies penalise efficient producers in other sectors and reduce employment opportunities and economic activity in other areas of the economy. The diversion of factors of production into inefficient industries and away from competitive industries acts as a constraint to medium and long-term economic development. The diversion of Australia's resources (land, labour and capital) into biofuel production and away from other areas of production potentially reduces Australia's international competitiveness - particularly over the medium to long term - resulting in reduced output and value-added across a wide range of industries, including agriculture.

It has been argued that a subsidised biofuel industry would compete directly and unfairly with other industries using the same inputs. By artificially increasing demand for inputs (through a subsidy for one sector), prices may rise or supply may be reduced. In other words, support for biofuels may lead to increases in input costs for other regionally based industries. For example, intensive livestock industries (like pork, poultry and grain fed beef) and other users (such as food producers) currently use potential biofuel feedstocks, such as grains and molasses. Producers in the livestock industry have argued that subsidising grain-based ethanol will distort the domestic feedgrain market to the detriment of all grain-dependent livestock industries, including those using grain for drought feeding. The biofuel industry would result in an increase in prices paid by or for these products.

There is little quantitative evidence available identifying the extent that biofuels will compete with other users for inputs, particularly feedstocks. However, as producers and consumers of grains and sugar products in Australia face world prices for these products, it is unlikely that any increase in domestic consumption following an expansion of the biofuels industry will result in a change in the price of the good. Rather, it is likely that increased consumption will be sourced by diverting product away from exports towards domestic consumption. In the CARE 2001 study (prepared for the Australian Biofuels Association) it was found that there was potential for further ethanol production without supply shortages by diverting molasses from export markets to local processing and that the livestock feed industry would be largely unaffected (CARE, 2001, p. 14).

It is worth noting that in a recent study into the feedgrain market by ABARE, it was projected that between 2003-2004 and 2007-2008, the number of livestock on feed would increase substantially. The largest increases in livestock numbers are projected in the cattle feedlot industry (up 25% in the five years), poultry meat production (up 15%) and pork production (up 13%). As a result, domestic consumption of feedgrains is expected to rise nearly 20% from 8.4 Mt to 10.0 Mt. However, it is projected that in 2007-2008, approximately 11.0 Mt of grain would still be exported. An important finding in that study was that the regional pattern of grain transfers within Australia is likely to change. As a result, some regions may face increased costs of using grains associated with transport costs due to the need to import grains from different regions (Hafi and Connell, 2003). The extent to which increased demand for grain associated with increased output in the biofuels industry may compound this is unclear.

In choosing to subsidise one industry over another (that is, discriminate between businesses competing for common inputs), the broader impacts must be considered as there is a considerable risk that existing employment and economic activity in other industries will be displaced. There will also be flow-on impacts to other businesses in the communities affected.

Another potential risk of supporting biofuel production is that it can lead to the promotion of unsustainable development with industries collapsing or leaving a region once the assistance lapses. Providing assistance to specific industries can result in activity that is not well suited to a region’s natural advantages. If the biofuel industry is not viable in the long term then any regional development and employment benefits are unsustainable and structural adjustment assistance by governments may be required when the industry is no longer able to operate. As a result, there is a risk that providing support for new biofuel plants on regional development grounds may be counterproductive, with short-term benefits being outweighed by the long-term structural needs of communities that would result from plant closures. Alternatively, the ‘footloose’ nature of some recipients can result in firms and industries leaving a region once the support lapses.

Similarly, diversification of crops and industries can provide greater stability and resilience to shocks for farmers and communities, but it can also mean that the benefits of high growth are not captured. Recent research by the BTRE indicates that regions with a highly diverse industry structure tend to experience more stable economic performance; however, the analysis does not support the claim that a highly diverse industry structure is associated with greater regional growth prospects (BTRE, 2003b, p. 48).

Subsidising biofuel production may be an expensive approach to achieving regional development policy goals. Biofuel production is not spread broadly through regional communities and it may encourage communities to engage in activities without any competitive or comparative advantage. In fact, biofuel production is not likely to be confined to regional areas - Chapter 7 showed that the most viable biodiesel production is from waste cooking oil, which is likely to be located in urban areas. As a result, direct assistance to this particular industry may be an indiscriminate and inefficient approach to achieving regional development policy goals.

In one of the rare pieces of literature attempting to quantify the costs associated with biofuel production, the Frontier Centre for Public Policy in Canada found that: *"The best argument for ethanol, that it diversifies rural economies and makes them more sustainable fades when you consider .... each ethanol job created in Minnedosa costs about $75,000 in taxpayer subsidies"* (Sopuck, 2002, p. 3).

A rough estimate of the equivalent government expenditure per biofuel job created for Australia is attempted in the next section.

The Canadian estimate is based on analysis of an ethanol plant in the town of Minnedosa, which is located in the Canadian province of Manitoba. In Manitoba, ethanol production is supported by a 2.5c/L tax rebate. They produce 90 ML of ethanol blended gasoline which equates therefore to a subsidy of $2.25m (Canadian) a year. The study found that subsidised expansion of ethanol will harm Manitoba's successful livestock industry by raising feedgrain costs. The Frontier Centre stated: *"Subsidised value-added businesses are vulnerable to changes in government policy, budgetary constraints and can be blindsided by price declines often brought about by over-supply….the problem with subsidies is that they distort the marketplace, raising input prices and creating more supply and lower prices"* (Sopuck, 2002, p. 6).

The study concluded that mandating and subsidising ethanol-blended gasoline in Manitoba would have the perverse effect of harming rural economic diversification and development (Sopuck, 2002, p. 10). Interestingly, the report also noted that: *"if environmental improvement is a public policy goal ... then ... would be far better off spending ... on projects that directly improve the environment"* (Sopuck, 2002, p. 9).

14.4 Government Expenditure per Biofuel Job in Australia

Expansion of the biofuel industry will result in increased jobs within that industry. However, as discussed above, it is likely that many of those jobs will come at the expense of jobs in other sectors or regions. That is, the increase in job numbers in the biofuel industry reported in studies does not represent the total increase in employment in Australia, which is likely to be substantially lower.

In the case where expansion of biofuel production can only occur with government assistance, an estimate of government expenditure per biofuel job in Australia is provided below. In the next chapter a full economic cost (in terms of lower GDP) is calculated. The analysis in this chapter focuses on the 350 ML biofuel target and calculates the cost to government per biofuels job of achieving that target. As discussed in Chapter 11, to reach the 350 ML target, an extra 235 ML of biofuel would need to be produced annually (assuming production of 115 ML in the reference no policy change case).

Table 75 provides estimates of the government expenditure required per biofuel job for the case where assistance is provided to all biofuel production in 2010 (that is, the entire 350 ML). The second case is where assistance is provided only to the production of biofuels over and above that which is projected to occur in the reference case (that is, to only 235 ML). The minimum assistance estimates are based on the viability analysis in Chapter 7 and the economic analysis that follows in Chapter 15. According to these analyses, the minimum government assistance required to induce sufficient investment to meet the 350 ML target (that is, the additional 235 ML in 2010) is approximately $30m in 2010 (in 2003 dollars). Alternatively, if the assistance were provided to all biofuels production, this is a transfer of approximately $44m.

Estimates of the number of jobs created, related to the operation of the biofuel industry, vary. However, as noted earlier there is some consistency in the literature that around 36 direct jobs are generated per plant. Despite the concerns previously expressed about the limitations of these estimates, in the absence of more robust figures, this estimate is used in this analysis. The remaining assumption required is the number of new plants likely to be operating to fulfil the 350 ML target. Evidence on this is less clear. The viability analysis in Chapter 7 indicated there could be around 6 plants of varying sizes. The employment estimates relate to a 60 ML plant. As a result, for the purposes of this analysis it is assumed that the additional production would require the equivalent of around 4 new plants. In reality, rather than four plants of similar capacity, there is likely to be a mix of smaller and larger plants.

Based on these assumptions, and assuming a multiplier effect of around 2 it is estimated that government expenditure per job created in biofuel and related industries (including both direct and indirect) is around $70,000 in 2010 to produce the extra 235 ML or $101,000 if the assistance is applied to the whole 350 ML. If the indirect impacts are ignored and only the direct jobs created in biofuel plants are examined the government expenditure per job rises to around $210,000 in 2010 to produce the extra 235 ML or $303,000 if the assistance is applied to the whole 350 ML.

Table 75. Government expenditure per job

	Case 1: Subsidise all production	Case 2: Subsidise additional production
Biofuel production	350 ML	235 ML
Government expenditure	$43.6m	$30.2m
Total employment associated with biofuel production	432	432
- No of plants	4	4
- direct jobs per plant	36	36
- total direct employment	144	144
- indirect jobs per plant (assuming a multiplier of 2)	72	72
Expenditure per job (direct and indirect)	$100,926	$69,907
Expenditure per direct job	$302,778	$209,722

It is important to remember that the employment numbers are all based on ethanol not biodiesel. The additional production of biodiesel, accounting for around 60 ML (the equivalent of around 1 plant) of the target would be based on waste cooking oil located in urban areas close to the feedstock source. As a result, this component is unlikely to be related to regional development goals.

The key message to arise from this analysis is that government expenditure to support a biofuels industry on regional development (i.e. employment) grounds alone involves significant government expenditure. It is also important to bear in mind that these jobs are likely to involve few, if any, net new jobs for the Australian economy.

14.5 Potential Locations for Biofuel Plants

It is generally accepted that as the costs of transporting raw feedstock are generally higher than the cost of transporting biofuels, biofuels plants are generally located close to feedstock supplies. For biofuels based on agricultural feedstock inputs this means there is a high likelihood that most of these plants will be located in rural areas. However, for biofuels based on feedstock such as waste cooking oils, the supply of which will be predominantly located in urban areas, biofuel plants will likely be constructed in urban locations. While biofuel production may tend to be concentrated more in rural areas, the consumption of biofuels will be predominantly in metropolitan locations.

The viability analysis in Chapter 7 indicated that C molasses, waste starch, waste cooking oil and sorghum/feed wheat were the most viable potential sources of feedstock for biofuel production in Australia. As a result, rural sugar growing areas in northern Queensland and grain growing areas in Queensland and northern and southern NSW are the most likely locations for biofuel plants. For example, the areas surrounding Cairns, Townsville, Mackay and Bundaberg in Queensland are prominent sugarcane areas. In NSW, grain is a major industry in the wheat belt towns such as Dubbo, Forbes, Parkes, Gilgandra, Coonabarabran, Moree, Gunnedah, Narrabri, Tamworth, Junee and Condobolin.

During the course of this study, information on the proposed location of plants was gathered and a number of biofuel proponents provided information regarding potential locations of facilities. Existing facilities are understood to be in:

- Nowra (southern NSW) using waste starch (Manildra);
- Sarina (northern Queensland) using C molasses (CSR);
- Moama (southern NSW) using canola (Biofuel Australia); and
- Maitland (Rutherford near Newcastle, NSW) using cooking oil (Biodiesel Industries Australia). This plant was only recently opened in March 2003 (Macfarlane, 12 March 2003).

Locations proposed for new ethanol plants are all in regional Australia and include: Dalby (Qld), Burdekin (Qld), Gunnedah (NSW), Mossman (Qld), Coleambally (NSW), Swan Hill (Vic), Forbes (NSW), Western Australia (possibly Kwinana) and many more in Queensland and NSW.

Locations proposed for new biodiesel plants include: Newcastle (NSW), Melbourne (Footscray, Vic), Beaudesert (Qld), Deniliquin (NSW), Brisbane (Qld), Sydney (NSW), Melbourne (Vic), Bunbury (WA), Albury-Wodonga (NSW-Vic) and Millicent (SA). This list reinforces the view that most biodiesel plants will be based in metropolitan or urban locations close to the largest supply of waste cooking oil.

The regional impact of a biofuel facility in each of these and other locations will vary depending on many interrelated factors including the:

- feedstock type (source and cost determines technology, labour and capital requirements etc);
- size of plant (generally, the larger the plant, the larger the impact);
- other plant features (e.g. the existence of different by-products, such as high protein meals or fertilisers which create differing employment impacts and other linkages within the regional economies);
- technology used (e.g. capital intensiveness, whether it is local or imported or if specialist skills are required to maintain it);
- size of town/region (larger towns/regions tend to be more diverse, able to supply more inputs locally and therefore capture more flow-on benefits);
- existing regional industry structure / economic base (less diversified towns/regions will tend to need more imported inputs and services and therefore receive smaller flow-on impacts);
- extent to which plant inputs can be locally sourced rather than imported from another region or from overseas (both capital and labour, in particular, the extent to which specialist skills unavailable in many regions might be required);
- transport links and infrastructure (the availability and quality of both infrastructure and operators to transport raw materials and products via roads and rail);
- other infrastructure (the availability and quality of gas, electricity, water, etc);
- access/distances to farmers/feedstock supplies and to major metropolitan markets, and
- existence and extent of any unemployed labour (determines whether the impact is actually new employment versus a transfer of existing employment from elsewhere).

As a result, each plant location will involve a unique combination of these factors, which will determine the specific regional impact of any proposed biofuel plant. Employment effects are different in different parts of the country because all of these factors influence the size of the multiplier used to estimate flow-on impacts (Hillring, 2002, pp. 445-446). In other words, the regional impacts of these plants are location-specific and given the developmental and commercial-in-confidence nature of most biofuels plant proposals, this study is not able to extend this analysis to assess the possible regional impacts of particular plants proposed for particular locations.

14.6 Conclusions

The major potential benefits of supporting a biofuels industry can be summarised as:

- strengthened regional economic opportunities (increased employment, investment, income, output etc);
- greater regional capability to withstand structural change (due to a more diverse economic base); and
- improved social outcomes (such as increased community confidence, social cohesion, social capital, etc).

Overall, there are regional benefits to be gained from some forms of biofuels production plants (ethanol); however these would come at a considerable cost to government and the Australian taxpayer. The estimates of direct employment impacts are reasonably consistent at around 30-40 jobs per plant. The estimates of indirect flow-on employment impacts were also generally similar in the range of 180 to 200 jobs. Construction employment varied widely from around 60 to 500. Multipliers used to estimate indirect impacts varied considerably from 2 to 60 but were most commonly around the 5 to 6 range. Government expenditure per direct job in 2010 is estimated to be between \$210,000 and \$303,000, with the expenditure per both direct and indirect jobs estimated to be in the range \$70,000 to \$101,000.

There are, however, some important limitations to the benefit estimates, including:

- lack of analysis of net national impacts;
- the extent to which benefits are new employment versus displacing existing employment in other industries or locations;
- use of very high multipliers to estimate flow-on benefits; and
- lack of transferability of results given the location-specific nature of biofuel plants.

Overall, the impact of construction of new biofuels plants on regions will vary according to many factors (including the source and cost of feedstock, differences in plant size, infrastructure and the economic structure of the regional economy). As a result, the net regional impact for the whole of Australia cannot be practically quantified.

It is likely that some regions would gain significant economic benefits from government support of a biofuels industry; however, when a net national perspective is taken other regions are also likely to suffer economic losses as a result. The net national impact is uncertain. The estimates in the literature probably overestimate the extent of the regional benefits a biofuel industry would bring to regional Australia.

15 ECONOMIC IMPLICATIONS OF THE 350 ML BIOFUELS TARGET

In this section the national economic impact associated with achieving a 350 ML biofuel target were assessed. ABARE's Australian Trade and Environment Model (AUSTEM) was used to estimate the benefits and costs associated with meeting the target. AUSTEM is a multi-sector dynamic general equilibrium model of the Australian economy (see Box 2).

To estimate net economic benefits to Australia associated with meeting the biofuels target, it is necessary to compare what is likely to happen with output under current policy settings (the reduction in production grants from 2008, and the provision of a capital subsidy), known as the reference case, with the pattern of output and investment that would be needed to meet the target. The reference case is described in more detail in Chapter 11.

In the reference case, fuel ethanol production is assumed to reach 85 ML and biodiesel production 30 ML by 2010. In order to reach the target of 350 ML, a subsidy to biofuels production is required. The level of subsidy (to ethanol and biodiesel production) modelled in AUSTEM is based on the viability analysis presented in Chapter 7. Table 76 draws together information from tables 6, 8 and 9 regarding the threshold level of oil based fuel prices and the revenue required to make ethanol produced from C molasses or cereal grains, and biodiesel produced from waste cooking oil economically viable.

On a biofuel energy equivalent basis, a subsidy of 14c/L and 5c/l is required for ethanol and biodiesel respectively. However, these subsidy levels need to be scaled by the appropriate energy content to enable them to be compared with petrol on a volume basis. With the energy content of a litre of fuel ethanol at 68% that of gasoline, and the energy content of biodiesel approximately 90% that of diesel, the subsidy required on a volumetric basis is 21c/l and 6c/l for ethanol and biodiesel respectively (see Table 76).

With the subsidy applied to new investment only, the total value of the subsidy to the biofuels industry in 2010 (in ethanol and biodiesel equivalent terms) is estimated to be approximately $31m (in 2003 dollars). This will induce sufficient investment to produce an additional 205 ML of ethanol and 30 ML of biodiesel, relative to the reference case. In order to ensure investment occurs in the manner necessary to meet the target by 2010, the required subsidy is assumed to be ongoing post 2010.

Table 76. Calculation of required subsidies[a]

	c/L (in petrol equivalent terms)	**c/L[b]** (in ethanol equivalent terms)
Ethanol		
Estimated medium term ex refinery price	**29**	**20**
Net required revenue (C molasses)	48	33
Subsidy	21	14
	c/L (in diesel equivalent terms)	**c/L** (in biodiesel equivalent terms)
Biodiesel		
Estimated medium term ex refinery price	**33**	**30**
Net required revenue (Waste cooking oil)	39	35
Subsidy	6	5

[a] Totals may not appear to add due to rounding.

[b] The energy content (energy density) of a litre of fuel ethanol is 68% that of gasoline. The energy content of biodiesel is assumed to be 90% that of diesel.

Box 2: Australian Trade and Environment Model (AUSTEM).

AUSTEM is a multisector dynamic general equilibrium model of the Australian economy and was developed by ABARE to address economic and policy issues of importance to Australia.

AUSTEM is suitable for analysing the impact of increasing biofuels production because of its detailed coverage of commodities and sectors, including the transport fuel sector. In particular:

- It provides a complete accounting depiction of the Australian economy — i.e. it embodies all flows of funds in the Australian economy and allocates these flows according to sources and uses;
- It incorporates linkages between industries and between broad sectors of the economy — including the household sector, the government sector, the financial sector, and the external sector;
- For this analysis, it has been modified to depict separately the ethanol and biodiesel industries incorporating their linkages to other sectors of the economy — including linkages with their feedstock industries and interactions with the government budget. Government budget linkages also include current and proposed excise (and excise rebate) arrangements; and
- It is a state of the art general equilibrium model incorporating a range of advanced and innovative design characteristics.

15.1 Estimated GDP Effects

As a result of the need to subsidise the biofuels industry to reach the target, annual gross domestic product (GDP) for the Australian economy in 2010 is estimated to be $70.9m (in 2003 dollars) lower relative to the reference case.

This economic loss arises from three main sources. First, using transport fuels that are more costly to produce (such as ethanol and biodiesel) reduces economic efficiency. Second, increasing output in the biofuels industry requires resources (such as labour and capital) to be attracted away from other economically productive activities, thus reducing the total value of output in other sectors of the economy. Finally, there is a need to fund the subsidy (via increased taxes or reduced government expenditure on services), which further decreases economic efficiency within the economy.

The loss in economic efficiency (often referred to as the deadweight loss) within the fuel transport sector of using biofuels instead of oil based transport fuels is estimated to be approximately $15m. The remaining $55.9m results from the reduced economic activity arising from both the diversion of factor resources away from other productive activities in the economy and the impact of funding the subsidy through raising additional taxation revenue. Additional tax revenue is required to offset the loss in revenue from the displaced petrol and diesel sales, as well as to fund the required subsidies to the biofuels manufacturers.

For example, the impact of lower resource availability and increased taxes is reduced production of goods and services available for exports and a fall in the value of exports relative to the reference case. Some of this change in the revenue reflects a diversion of grains and sugar away from the export market into feedstock for the ethanol industry, which reduces the value of grain and C molasses exports by $9.5m. Similarly, the substitution of biofuels for oil based transport fuels leads to an increase in the value of oil exports of $9.6m (in 2003 dollars) in 2010. In total, the impact is a total decline in export revenue in 2010 of $51.1m (in 2003) dollars relative to the reference case.

Increased competition for cereal grains may apply some upward pressure on domestic cereal grain prices, particularly in some regionalised markets (Hafi and Connell, 2003). However, given the additional cereal grains used to produce ethanol in this scenario are likely to be sourced mainly through reduced exports, any impact on prices in local cereal grain markets is likely to be minor.

Overall, in this scenario, the total net present value of the costs to the Australian economy (over the period 2004 to 2010) of meeting the target in 2010 is approximately $95m (in 2003 dollars) relative to the reference case.

In the case considered above, the subsidy has only been applied to additional production over and above that which is expected to occur in the reference case. If the subsidy were to be applied to output of the entire biofuels industry, GDP in 2010 would be $74.3m lower relative to the reference case. In this case, the total cost over the period of 2004 to 2010 is approximately $100m (in 2003 dollars) relative to the reference case.

15.2 Economic Costs per Biofuel Job Created

An estimate of the economic cost of each biofuel job created in meeting the 350 ML biofuel target is provided in Table 77. These estimates draw on the analysis of regional impacts (presented in Chapter 14) and the analysis of national economic impacts (presented earlier in this chapter).

In Table 77 the economic cost of both direct and indirect employment opportunities is estimated across two scenarios. First, the cost of each biofuel job is estimated for the case where assistance is provided to all biofuel production in 2010 (that is, the entire 350 ML). The second case is where assistance is provided only to the production of biofuels over and above that which is projected to occur in the reference case (that is, to only 235 ML).

In both cases, the number of new jobs created in biofuels and associated industries are the same. With approximately 4 new plants being required, total additional biofuel jobs created (both direct and indirect) is estimated to be 432 (see Chapter 14). Of course it goes without saying that this does not mean all these jobs are entirely new employment. As discussed in Chapter 14, labour, particularly skilled labour, is a mobile resource within the economy and development of a biofuels industry (assisted or not) would be expected to compete resources away from other sectors. This point is also made earlier in describing the potential sources of economic loss and the fact that increasing output in the biofuels industry requires resources (such as labour and capital) to be attracted away from other economically productive activities.

As described earlier, the economic cost of meeting the 350 ML target (in terms of lower GDP in 2010) is estimated to be between $70.9m and $74.3m (2003 dollars). Based on these results, it is estimated that the economic cost of each biofuels and related job (in 2010) is between $164,000 and $172,000. In the case of direct jobs created, the economic cost for each job in 2010 is estimated to be between $492,000 and $516,000 (Table 77).

Table 77. Economic costs per job created

	Case 1	Case 2
Biofuel production	350 ML	235 ML
Economic cost of meeting the 350 target in 2010	$74.3m	$70.9m
Total employment created in biofuels and associated industries (direct and indirect)	432	432
direct employment	144	144
indirect employment	288	288
Cost per job (direct and indirect)	$171,991	$164,120
Cost per direct job	$515,972	$492,361

15.3 Benefits of Avoided Health Costs and GHG Emissions Reductions

In order to obtain a measure of the full economic impact (or cost) of achieving the 350 ML target, it is also necessary to account for other possible economic costs or benefits that are not included in the AUSTEM analysis. An important example of this is the potential benefit associated with reduced emissions from burning fossil fuels. The benefits from reducing emissions from burning fossil fuels can be grouped into two categories: local health and environmental benefits and the global benefits of avoiding the damage associated with climate change.

15.3.1 Avoided Health Costs

In the case of the former, epidemiological studies have shown a link between concentrations of toxic substances in urban air sheds and morbidity and mortality rates amongst residents. Hence, benefits in the form of avoided health costs result from lower emissions of pollutants both at the tailpipe (i.e. directly associated with vehicle use) and further upstream (i.e. associated with fuel production and refining). Estimates of the economic benefit of avoided health costs (derived using value of life estimates) were described in detail in Chapter 11.

The potential saving in total health costs of meeting the 350 ML biofuels target in 2010 is estimated to be \$3.3m (Table 66). That is, health costs in 2010 in the 350 ML target case are estimated to be \$3.3m (in 2003 dollars) lower than in the reference case.

15.3.2 GHG Emissions Reductions

The additional biofuels consumption, necessary to obtain 350 ML in 2010, would result in a reduction in total greenhouse emissions in 2010 of approximately 268,000 tonnes. This reduction in greenhouse emissions is estimated to comprise 184,000 tonnes (from increased use of ethanol) and 84,000 tonnes (from increased use of biodiesel). In terms of the cost of greenhouse gas emission reductions, the estimated cost to government (in 2010) is estimated to be between \$113 and \$163/tonne CO_2-e (in 2003 dollars). The total economic cost associated with the reduction in emissions (again in 2010) is estimated to be between \$265 to \$277/tonne CO_2-e (in 2003 dollars).

The benefits that flow to Australia through the mitigation of climate change come in the form of reduced potential economic and environmental damages.

For all practical purposes, the present value of the economic benefits that flow to Australia from the mitigation of around 268,000 tonnes (CO_2-e) of greenhouse gas emission in 2010, can be taken to be immeasurably small. This is so because first, the contribution (of 268,000 tonnes) to global concentrations of greenhouse gases in the atmosphere is very small; second, the benefits that accrue do so in the distant future; and third, the benefits are shared throughout the world, with some countries likely to have a greater capacity to gain than Australia.

However, the present value of the benefits that flow to Australia through the mitigation of climate change are likely to represent some positive amount if a workable global resolution to the climate change problem is achieved and if emission rights were to become tradeable assets. In this case the emissions avoided in each year can be valued positively. In the absence, at the present time, of an international market value for carbon dioxide equivalent emissions, the Australian Greenhouse Office has suggested use, on a illustrative basis, of the values contained in the 1999 publication, Discussion Paper 2 – Issuing the Permits (AGO, 1999b). The discussion paper postulated a permit price range of \$10 to \$50 a tonne CO_2-e. The lower value of \$10 a tonne CO_2-e is consistent with the upper bound of the cost to government of abatement purchased under round 1 of the Greenhouse Gas Abatement Program (GGAP). The abatement purchased under GGAP relates to the period 2008-2012, which is the first commitment period under the Kyoto Protocol.

Taking this estimate of the price of emission permits as a measure of the benefits of a unit of greenhouse abatement, at the margin, the total value of the abatement associated with the increased use of biofuel is estimated by multiplying the quantity of emissions avoided by the forecast unit price of emission permits. For example, at $10 a tonne CO_2-e, the implied value of the benefits associated with the greenhouse gas abatement achieved from the additional 235 ML of biofuels used in 2010 is $2.7 million (in 2003 dollars), or 1.1c/L.

A comprehensive assessment of the environmental impact of meeting a 350 ML biofuels target was presented in Chapter 10 and Chapter 11.

16 CONCLUSIONS

This study has assessed the viability, in technical and economic terms, of producing biofuels in Australia. Importantly, it has not looked closely at the commercial viability of biofuels (for example, at factors such as commercially required rates of return), which will also have an impact on actual outcomes.

With this caution in mind, ethanol from waste starch and biodiesel from waste cooking oil both appear to be (or are close to being) viable without ongoing government financial assistance. These fuel options should be able to compete effectively in an environment where they are taxed on a comparable basis with other fuels. However, in both cases, future growth is expected to be modest due to the limited availability of feedstock supplies.

Ethanol from molasses and whole cereal grains and biodiesel from tallow or oilseeds all require substantial and ongoing government assistance to be economically viable.

Changes in fuel standards, both those currently being implemented and those planned for the period after 2006, are unlikely to have a significant impact on the viability of ethanol. A small increase at most is likely in the threshold price of petrol, with which ethanol must compete, in response to an expected requirement for production of higher octane fuel. Moreover, the addition of ethanol to petrol may not be the most cost-effective option for the refining industry in producing higher octane fuel (and indeed would not on its own achieve this, without some additional refining). Prospective fuel standards are unlikely to impact the future viability of biodiesel.

Particularly with the prospect of significantly cleaner petrol and diesel in use in the vehicle fleet by 2010, the net environmental impacts of increased biofuels, while positive, are small, in overall terms. While this conclusion is unlikely to alter with better information, more knowledge is nevertheless needed about the impact of evaporative emissions from E10, to make a clear judgement on the impact of E10 on ozone formation.

Regional benefits of increased biofuels use would occur in locations with new or expanded ethanol plants. This is likely to be in parts of Queensland and New South Wales. While there would be positive employment impacts, not all of this employment would be additional, in a national or possibly a regional context, given the mobility of and alternative demand for skilled labour in the economy.

The costs of implementing a policy of assisting the Australian biofuels industry to meet a 350 ML biofuels target are estimated to exceed the benefits. This conclusion is based, firstly, on comparison of the estimate of the required subsidy in energy equivalent terms (14c/L of ethanol and 5c/L of biodiesel), with dollar value estimates of the beneficial but small health impact from increased biofuels use (1.4c/L, increasing to 2.2c/L, if the additional ethanol were produced using co-generated energy). Secondly, it reflects the high cost per tonne of reductions in carbon dioxide equivalent emissions, compared with the cost to government of abatement purchased under the Greenhouse Gas Abatement Program.

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Australian Government

APPROPRIATENESS OF A 350 MILLION LITRE BIOFUELS TARGET

APPENDICES

DECEMBER 2003

REPORT TO THE AUSTRALIAN GOVERNMENT
DEPARTMENT OF INDUSTRY TOURISM AND RESOURCES

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APPENDIX I. LITERATURE REVIEW - BIOFUELS TAILPIPE EMISSIONS

INTRODUCTION

Part of the scope of this review is to examine existing local and international literature, studies and scientific reports to assess the net environmental (greenhouse, air quality and other) impact of replacing fossil fuels with biofuels, produced in Australia from renewable resources, in the Australian transport mix. Where possible, studies applicable to Australian conditions were targeted. The examination of the emissions associated with agricultural production of crops and fuel production were restricted to accounting in the life cycle analyses, and environmental issues specific to these processes were not examined in this review (e.g. land/water degradation due to agricultural production). Where possible, original publications cited in references were sourced.

The biofuels considered were biodiesel for heavy duty vehicles (as 100%, 20% or 5% blend with diesel) and 10% (v/v) ethanol in conventional petrol (E10 or "petrohol") for passenger vehicles. The biofuel or ethanol can be sourced from a variety of crops.

This part of the review is focussed specifically on tailpipe emissions. Issues such as biofuel effect on vehicle performance (driveability) and engine and fuel component durability (corrosion, permeability) were beyond the scope of this review. These issues are discussed in several publications for ethanol (including: Orbital Engine Company, 2002; Orbital, 2003; API, 2001; NSTC, 1997, and Arters *et al.,* 2002) and biodiesel (including: The National Biodiesel Board web site and www.biodiesel.org/fleets/summary.htm)

The potential environmental impact of biofuels on groundwater contamination (via leaching) is not covered in this review due to the anticipated negligible effect expected as a result of the biofuel components of the fuel blends, considered here, being readily biodegradable. A negative impact of this could be the preferential bacterial biodegradation of ethanol over other toxic fuel compounds. Also, at high concentrations ethanol is toxic to microbes. With respect to ethanol specifically, groundwater contamination is possible for all oxygenate compounds; however, ethanol has a very short half-life of 4.1 days and is readily biodegradable. The banning of MTBE (half-life > 120 days) as a fuel oxygenate is being considered in the USA due to MTBE contamination of groundwater, which is primarily related to inadequate fuel transport and storage regimes. A potential adverse impact of ethanol-petrol fuel blends could be the co-solvency effect of ethanol with aromatic compounds (benzene, toluene, ethylbenzene, xylenes) in petrol resulting in increased solubility and hence mobility of these environmentally undesirable compounds (Environment Australia, 2002). It is, however, difficult to gauge the comparative environmental impacts from storage tank leakages and spillages of the toxic compounds present in the conventional petrol component of E10, relative to conventional petrol.

Biodiesel

Biodiesel is a generic name for fuels obtained by the trans-esterification of vegetable (or animal) oils. This produces a fuel with very similar combustion properties to pure diesel, but with lower viscosity. Often the term biodiesel refers to rapeseed oil methylester (RME), the main European biodiesel. In the USA the main biodiesel is produced from soybean oil, called soydiesel.

Potential sources of biodiesel, particularly relevant to Australia, are: canola, soybean, rape, tallow and waste cooking oils. Methanol or ethanol can be used for the esterification process, though the process is simpler, and proceeds more rapidly, when methanol is used. Biodiesel can be used in a diesel engine without modification.

The fuel consumption of biodiesel per kilometre travelled is similar to that of diesel when biodiesel is used as a diesel blend. Biodiesel has a lower energy content than diesel. Thus there is increased fuel consumption when pure biodiesel is used (Taberski *et al*., 1999).

The advantages of biodiesel are:

- 100% biodiesel contains almost no sulfur and no aromatics that, in a properly tuned engine, should lead to decreased PM exhaust emissions;
- the absence of sulfur allows more effective use of oxidation catalysts;
- it is bio-degradable and non-toxic; and
- in a mixture with low-sulfur diesel, biodiesel can improver lubricity (Arcoumanis, 2000).

Potential issues or disadvantages associated with biodiesel such as: effect on driveability, oxidation stability, corrosion, standardisation of biodiesel production, higher kinematic viscosity of biodiesel, dilution of engine lubricant oil and possible refuelling infrastructure required to handle biodiesels were considered beyond the scope of this literature review.

Petrohol (E10)

Ethanol is a polar, small chain, alcohol molecule (C_2H_5OH), is totally water-soluble and contains 34.7% (wt) oxygen. Ethanol is significantly different from the petrol containing HC regarding a number of properties: heat of vaporisation; calorific content (heating value); boiling point; vapour pressure; flammability; viscosity; flash point, and anti-knock performance (octane rating). At low moisture levels ethanol in fuel could be beneficial in removing moisture contamination of fuel; however, phase separation would occur at higher water concentrations due to the high solubility of ethanol in water. In addition, ethanol in fuel is corrosive due to the electrical conductivity of ethanol (unblended petrol is an electrical insulator). As a result, material selection may be an issue although this would be expected to be only significant for high concentrations of ethanol in fuel. High concentrations of ethanol can degrade other materials such as rubber and can increase fuel permeability of rubber and elastomer components in vehicle fuel delivery systems. Although pure ethanol is poisonous, it is less acutely toxic than the aromatic compounds present in fuel. It is bio-degradable and would be expected to be tolerated by many organisms, at least at moderate concentrations.

The addition of 10% (v/v) ethanol to conventional fuel to produce petrohol (or E10) can be considered as a *fuel substitute* (fossil fuels replacement) or as a *fuel oxygenate additive*. Oxygenates are primarily used in fuel to reduce CO emissions (to reduce the impact of winter inversions). In 1990 amendments to the US Clean Air Act established the Federal Reformulated Gasoline (RFG) Program, whereby RFG fuel must contain 2% oxygen (w/w) and is required by law in certain regions of the US. In areas such as California more stringent requirements are in place, whereby the required oxygen content is 2.7% (w/w). In the US, particularly in the corn producing states of the mid west, ethanol is widely used as an oxygenate additive as 8% in “oxyfuel” and 6% in RFG. The US congress has introduced a subsidy for ethanol use in gasoline, which has sometimes resulted in ethanol fuel contents at 10%.

The chemical and physical effects of adding ethanol to conventional fuel are well understood (API, 2001). A comprehensive inter-agency US assessment of the impact of oxygenates in fuels was conducted in 1997 (NSTC, 1997). A number of studies have found that effects of oxygenates on fuel/emission properties is independent of the type or chemical structure of oxygenate compound used (Reuter *et al*., 1992; Hood and Farina, 1995). The blending of ethanol (and other oxygenates) in petrol also improves the anti-knock or octane quality of the fuel. This is due mainly to charge cooling within the combustion chamber from the high latent heat of vaporisation characteristic of ethanol. E10 has approximately 20% higher latent heat of vaporisation than petrol, producing the same net combustion energy (API, 2001).

Combustion

Fuel combustion in motor vehicles is a highly complex process, with many variables influencing tailpipe emissions. One of the most significant variables is the effect of the air to fuel ratio (AFR) in the engine. A number of other variables can also have major impacts including: vehicle/engine technology; condition of vehicle (maintenance); driving conditions; as well as the type and condition of pollution control systems (if present). The effects of biofuels on emissions can depend on all of these variables. The only comprehensive method to assess the effect of biofuels would be to test each combination of variables, with each fuel type, the scale of which makes this an impractical task.

The vast majority of vehicle emission data currently available for model development are collected through laboratory testing programs, where vehicles are tested on chassis dynamometers using driving cycles designed to reflect on-road activities. Insufficient data are available accurately to reflect the on-road emission rates from all types of vehicles (new models and older vehicles that have continued to age). Furthermore, concerns are significant regarding the applicability of laboratory emissions data to accurately reflect on-road emissions.

Test results can indicate both high variances (within vehicle and between vehicles) for the same test condition, as well as different tests and fuels. Hence, large amounts of data are required to develop reliable emission rate models that can identify the changes due to alternative fuels.

It is difficult to determine if the sample of vehicles used is representative for the current Australian fleet and, therefore, whether or not quantifying their emissions will bias the emissions modelling.

The combustion studies were prioritised according to the following criteria:

- applicability to Australian conditions;
- representative of in-service vehicle fleet (Australia);
- representative vehicle drive cycles;
- statistically valid methods used including a sample size sufficient to allow statistical analysis,
- comprehensive emissions testing and analysis; and
- current relevance.

No study was found that satisfied all of these requirements. In Australia very limited studies have been attempted to assess the effects of E10 and biodiesel. Studies that have tested the vehicle combustion of biofuels generally restrict tailpipe emission measurement to the regulated pollutants (CO, HC, NO_x). Reliable data, in particular, for GHG emissions, PM and air toxics is generally unavailable. An Australian in-service vehicle emission (tailpipe and evaporative) study, combined with an air quality monitoring program (including OFP and health risk assessment), would be required to assess the impacts of biofuels more effectively.

Petrohol (E10)

A major focus of this review is aimed at the effect on tailpipe emissions of adding ethanol in conventional petrol.

Ethanol has been used as a fuel extender in Australia for a considerable time and is normally used as E10 (also E20) with the particular aim of producing PULP (95 RON). In Australia there have been a number of ethanol fuel blend trials — a large number of reports were produced for the National Energy Research Development and Demonstration Council (NERDDC). Of these one was a field trial conducted in 1980-83 of 100 vehicles and this reported the E15 impacts on fuel parameters, engines and vehicle fuel delivery systems (CSR, 1983). The only other field trial conducted in Australia was also commissioned by the ERDC in 1994 (APACE, 1998), reporting on the impacts of E10 on vehicle emissions, as well as the effects on a range of fuel, vehicle, component-compatibility and water tolerance issues. The data set of this study contains the only reliable emission results of E10 in Australia and is presented later in this review. A recent field trial initiated by the Department of

Environment and Heritage (then known as Environment Australia) aimed at determining the impacts of E20 on the Australian vehicle fleet has also been conducted (Orbital, 2003). This study is also included in the review, despite the higher ethanol fuel concentration, due to its relevance to the Australian market.

Fuel composition determines both the exhaust and the evaporative emissions of HC from motor vehicles. An important emission reduction effect, from the addition of 10% ethanol, is achieved through dilution due to the unburned petrol HC component of tailpipe emissions.

Evaporative Emissions

Vehicle evaporative emissions are fuel (and oil) HC vapours that are released by the vehicle prior to combustion and contribute to the total downstream emissions of HC. The factors that effect evaporative vehicle emissions are: fuel volatility, ambient temperature, driving conditions, as well as vehicle specific factors such as vehicle design and maintenance, and type and condition of evaporative emission control systems (if fitted to the vehicle) (Environmental Protection Authority of Victoria, 1997). Due to the low volatility of diesel fuel, the evaporative emissions effects are limited to petrol.

Evaporative losses of HC originate from a variety of vehicle parts: fuel tank, permeation through fuel hoses, carbon canister, and carburettor bowl in older vehicles, and can be categorised as:

- diurnal losses (daytime heating of the fuel tank/lines mainly causing breakthrough of a saturated carbon canister);
- running losses (vaporisation of fuel during vehicle operation);
- hot soak losses (vapour loss after vehicle stops and cools down);
- resting losses (fuel permeation through rubber engine components and liquid fuel-leakage); and
- refuelling losses (vehicle refuelling and bulk tanker refilling).

With the exception of resting losses, all of the sources of evaporative emissions increase as a function of increasing fuel volatility and ambient temperature. Vehicles must comply with the relevant ADR for regulated evaporative emissions (HC). The ADR37/00 specifies fuel volatility in terms of RVP. Due to the low volatility (or RVP) of diesel fuels, evaporative emissions are not as significant as those from petrol fuelled vehicles.

An ABARE report (ABARE, 2001) stated that “it has been noted that in the near future, evaporative emissions from cars will comprise about half the total hydrocarbon emissions in a city’s air-shed.”.

Volatility Characteristics

The volatility characteristics that are important to vehicle emissions (as well as vehicle operation) are vapour pressure, distillation and vapour-liquid ratio (API, 2001). The addition of ethanol (and other alcohols) to petrol increases the vapour pressure of the blend, although not in proportion to concentration (or vapour pressure of the pure alcohol). The molecular polarity of alcohols (like water) causes the vapour pressure anomalies when blending ethanol with petrol. The vapour pressure of pure ethanol (2.3 psi / 15.9 kPa) is considerably lower relative to petrol (7-15 psi) due to the cohesive inter-molecular forces (hydrogen-bonding) present. Blending low concentrations of ethanol with petrol (non-polar), however, increases the blend vapour pressure due to the physical separation of the ethanol molecules and peaks at approximately 2-4% ethanol (v/v). The vapour pressure effect of ethanol is less in petrols of higher vapour pressures because the ethanol partial pressure is constant (for a given temperature) and is independent of the partial pressure of the petrol (API, 2001). The addition of 10% ethanol (v/v) to conventional petrol results in an increased RVP of approximately 1psi (6.9 kPa), relative to the conventional fuel and also changes the fuel distillation curve, in particular the range where 50% of the fuel will be vaporised.

Additional modification of the hydrocarbon content of the petrohol blends is required to compensate for the effect on fuel vapour pressure. Removing the C4 fraction and reducing the C5 can achieve this. This modification may, however, affect the fuel distillation curve and adversely affect vehicle driveability characteristics such as cold start driving (API, 2001).

In addition, new vapour pressure measurement techniques for alcohol fuel blends are required to replace the traditional RVP method, due to adverse water contamination effects (alcohols have high affinity for water).

Highly volatile fuels can cause vehicle vapour lock at elevated operating temperatures, causing increased evaporative emissions. The vapour/liquid volatility ratio is used to predict fuel performance in fuel delivery at high temperatures.

E10 Combustion (Downstream)

There is considerable uncertainty associated with tailpipe (downstream) vehicle emissions from E10 (and most other fuels) and the comparison of data between studies is subject to significant uncertainty. The majority of tailpipe emission studies are also restricted to the regulated pollutants (CO, NO_x, HC) and limited data is available concerning GHG emissions, PM or air toxics. The data is even more restricted with respect to the effects of E10.

The comparison of emissions between studies is limited by the: large number of variables associated with combustion; very small sample sizes commonly used; and the large variation of vehicle, engine and pollution control technologies used in individual studies.

A number of studies have examined tailpipe emissions using E10 (APACE, 1998; Andress, 2000; Leong *et al.,* 2002; Ragazzi, 1999; Howard, 1997; Mulawa *et al*., 1997a; Guerrieri *et al.,* 1995; Reuter, *et al*., 1992; Furey and King, 1980), the majority of which concluded that use of E10 resulted in decreased emissions of CO and, to a lesser degree HC (including benzene), combined with increased emissions of NO_x and acetaldehyde. An important issue related to the commonly observed reduction of CO and HC (combined with increased NO_x) is that it is a result of "enleanment" effect of the oxygen present in the ethanol molecule and that this effect is essentially negated with closed-loop control of engines. The effects on emissions of PM and various air toxics are not clear. One US study did conclude that PM emissions were lower using E10, relative to conventional fuel, based on a study using 10 in-use vehicles (Mulawa *et al*., 1997a). In addition, emissions of inorganic compounds such as Pb and SO_2, present in conventional fuel, would be expected to decrease with E10, due simply to volumetric dilution of the petrol.

In Australia ethanol has been used as a fuel extender for a considerable time and is normally used as E10 with the particular aim of producing PULP (95 RON). The only significant Australian vehicle emission study aimed at determining the effects of E10 on vehicle emissions was conducted by the NSW EPA for APACE Research (APACE, 1998), although non-CO_2 GHG emissions were not included. Non-CO_2 GHG emission factors were not determined in this study, however, reasonable estimates of CH_4 emissions can be derived from the CSIRO HC speciated study (component of the Petrohol study). The study showed (Table A1) that the effects of E10 on tailpipe emissions was a decrease in THC and CO, negligible or slight increase in NO_x and an increase in CO_2. The details of this study are outlined below.

Table A1. Tailpipe emissions (g/km) of regulated air pollutants from Pre- and Post-1986 vehicles with petrol and petrohol

	THC	CO	NO_x	CO_2	% change THC	% change CO	% change NO_x	% change CO_2
PRE-1986 Petrol	1.86	19.13	1.84	255.1				
PRE-1986 Petrohol	1.66	12.06	1.83	259.9	-10.8	-37.0	-0.5	+1.9
POST-1986 Petrol	0.66	8.45	1.39	264.4				
POST-1986 Petrohol	0.57	6.18	1.46	266.2	-13.6	-26.9	+5.0	+0.7

The most thorough overseas fuel oxygenates study was conducted by the US Auto/oil Air Quality Research Program (AQIRP), which commenced a comprehensive analysis of fuel oxygenates in a variety of fuel types in 1989. Two fleets of well maintained vehicles: "current" (10 pairs of 1989 model vehicles) and "older" (7 pairs of 1983-1985 model vehicles) technology were used. Federal test procedures were employed to measure exhaust and evaporative emissions (mass and speciated). E10, one of the fuels tested, was blended using 4 different base gasolines. One of the tests investigated the change in evaporative emissions when the RVP of an E10 blend was reduced by 6.9 kPa. Results from the AQIRP program have been summarised (API, 2001) and reported in a number of publications (AQIRP, 1993a, 1993b; Colucci and Wise, 1992, and Automotive Engineering, 1992a, 1992b).

Tailpipe emissions of a number of other criteria pollutants, such as sulfur and lead, would be expected to decrease with E10 due to the volumetric dilution of the ULP component in the fuel blend.

A number of studies have been conducted on tailpipe emissions of higher ethanol blends (Guerrieri *et al.*, 1995; Orbital, 2003, and others) and the findings largely correlate with the E10 emissions results. Guerrieri *et al.* (1995) tested 9 ethanol blends up to 40% ethanol in conventional petrol and found in general an approximately linear response in the emissions as ethanol content is increased.

Section 8.3.2 of the main report provides further information on the effects of E10 on emissions and air quality.

Biodiesel Combustion (Downstream)

Significant uncertainty is associated with the determination of tailpipe emissions, varying according to engine technology and condition, vehicle maintenance and also, if non-steady state test cycles are used, the accuracy with which the cycles have been performed by the test driver. Emission testing has been conducted with a wide range of vehicles utilisation a range of pollution control strategies. It is also difficult to compare the combustion emissions of substantially different fuels such as biodiesel and LSD.

Concerning the emissions from diesel vehicles, there are some generalisations associated with different fuels, which include:

- the less volatile and more aromatic the fuel, the higher the exhaust PM emissions;
- presence of sulfur in the fuel results in increased PM; and
- oxygenated fuels reduce PM due to more complete combustion, assuming that other fuel qualities (e.g. cetane number) remain constant.

In regard to fuel consumption, provided the fuel is within the normal specification range, then for a given engine technology and transport task, fuel economy will be related to the energy content of the fuel. The higher oxygen content and accompanied lower calorific content of biodiesel, relative to diesel, results in increased fuel consumption.

The extensive use of biodiesel fuels in the United States and Europe means that data is available on their emission characteristics during operational performance and this data was summarised by Beer *et al.* (2000). In Beer *et al.* (2001), more recent results, and some of the relevant older results were reviewed and the different studies were compared. US tailpipe data is presented on the Alternative Fuels Data Centre (AFDC) website and is covered in Beer *et al.* (2000).

There were discrepancies found between European and American biodiesel emissions results (Beer *et al.,* 2000, 2001). European data (Arcoumanis, 2000) suggested biodiesel gives a reduction in HC compared with LSD. CO tends to be lower for biodiesel, while NO_x tends to be slightly higher. PM may be lower (Buckmann and van Malsen, 1997) or it may be higher (Arcoumanis, 2000; Ceuterick and Spirinckx, 1999) but that is not clear. Within the variability and uncertainties associated with the fuels one should consider the particulate matter emissions of the two fuels to be much the same.

According to US results, using BD100 (soybean feedstock) in urban buses, relative to 500 ppm LSD tailpipe emissions of PM10 and CO were substantially reduced by 68% and 46% (g/km basis), respectively (Sheehan, *et al.*, 1998a, b).

Particulate Matter

Due to the absence of sulfur and aromatics, and the presence of oxygen in biodiesel, one would expect theoretically lower PM emissions. The impact of biodiesel on tailpipe emissions, in particular for PM, varies between studies. This was shown in the difference in PM emissions effect found in Stage 1 of the CSIRO comparison of transport fuels study (Beer *et al.,* 2000), which reported an increase in PM, and Stage 2 (Beer *et al.,* 2001), which reported a decrease in PM. The Stage 2 study used US Tier2 testing data from Sharp (Sharp *et al.,* 2000). This variability led the US EPA to use their statistical approach in “A Comprehensive Analysis of Biodiesel Impacts on Exhaust Emissions” (US EPA, 2002). Recent results by Sharp *et al.* (2000a, b) indicated that modern American engines show decreased PM emissions. Previous work by Motta *et al.* (1996) using biodiesel in an earlier generation of engines installed in buses, indicated higher PM emissions. Discrepancies in the PM emissions between studies may be related to whether the engine was optimised to run on biodiesel or diesel.

Biodiesel Effects on GHG emissions

There is a distinct lack of data for tailpipe emissions of the non-CO_2 GHG, particularly in Australia. Emission characteristics of N_2O and CH_4 have not been characterised; however, it would be expected that N_2O emissions should not be affected significantly.

Biodiesel Effects on Air Quality

The United States National Biodiesels Board summarised studies on the air toxics emitted during biodiesel combustion, compared to diesel combustion. These results are given on the web site (http://www.biodiesel.org/fleets/summary.shtml#attributes).

Tailpipe emission reductions of PM, CO and SO_2 are found for biodiesel compared to diesel and significantly emissions of PM smaller than 10 microns were 68% lower for buses run on biodiesel, relative to diesel (Sheehan *et al.,* 1998a, b). These reductions were in proportion to the amount of biodiesel in the fuel – for BD20 expect 20% emission reduction of BD100.

Beer *et al.* (2001) stated: *“There appear to be no health risks of air toxic emissions from biodiesel with respect to mortality, toxicity, fertility or teratology. All air toxic emissions from biodiesel are lower than equivalent diesel emissions except for acrolein. Though highly toxic, the slight increase in acrolein is offset by the decrease in the equally toxic aldehydes.”*

Beer *et al.* (2001) reviewed the air toxics tailpipe emissions from biodiesel combustion and concluded that relative to diesel, biodiesel resulted in:

- lower emission levels of most toxic and reactive HC species with only marginal increases found for acrolein, propionaldehyde, and hexanaldehyde,
- substantially decreased emissions of PAH and nitro-PAH compounds (30% with a catalytic converter, 12% without a catalytic converter) and virtually eliminated some of the heavier NPAH compounds,
- substantially reduced emissions of aldehydes and ketones,
- caused a dramatic change in the character of the heavier HC species, with only the esters that made up the biodiesel remaining in exhaust among the higher molecular weight HC; and
- the blending of biodiesel and diesel did not generate any new species not already present in diesel or biodiesel exhaust.

Biodiesel Effects on Ozone Forming Potential (OFP)

The significant negative impact on air quality is the increased tailpipe emissions of NO_x, which could infer an increased ozone forming impact and increased smog production propensity. The influence of biodiesel fuels, including rapeseed oil fuels, on the formation of photochemical smog, whose main component is ozone, may be inferred from the fact that ozone in Australian cities is mainly NO_x limited.

Beer *et al.* (2001) stated that there are reduced emissions of evaporative HC emissions (C_1 to C_{12} range) for biodiesel and the relative reactivity of speciated hydrocarbons with biodiesel was similar to that observed with diesel exhaust hydrocarbons. The speciated HC emissions from biodiesel, should therefore, result in a lower overall OFP than for speciated diesel hydrocarbons.

LITERATURE REVIEWED: PETROHOL (E10) COMBUSTION EMISSIONS

The primary source of emissions data for E10, for the purposes of the present study, is the APACE/NSW EPA Petrohol study (APACE, 1998).

A range of vehicle emission studies has been conducted around the world to assess the emission effects of adding ethanol to petrol. USA and Brazilian studies are the most common. A 16-vehicle (no catalyst/dual bed catalyst/TWC) emission study, using the Bangkok Driving Cycle test mode, was conducted in Thailand (Leong *et al.,* 2002) and compared E10 (and E15) with conventional ULP. The emissions tested were HC and air toxics and all vehicles showed reductions in HC combined with increased acetaldehyde and formaldehyde emissions with E10, relative to conventional ULP.

Engine dynamometer studies have also been conducted using E10 (and other ethanol concentration ranges) blends including a Korean study by Hsieh *et al.* (2003) and a Jordanian study by Al-Hasan (2003).

The majority of the studies found beneficial tailpipe emissions performance (not for aldehyde and NO_x emissions), using ethanol in a range of concentration ranges blended with conventional petrol. Evaporative emissions, however, are significantly effected by the addition of ethanol and the technical issues are worth considering.

Reuter *et al*. (1992) studied the effect of various oxygenates on vehicle emissions (FTP) using 20 vehicles (1989 model) and found a reduction in CO, HC and benzene emissions with all oxygenates (including ethanol), relative to the non-oxygenated fuels. The NO_x emissions increased for all oxygenates tested and acetaldehyde emissions increased for ethanol petrol blends.

The study of most relevance to vehicle emissions effects of E10 under Australian conditions is that conducted by the NSW EPA, in conjunction with APACE Research (APACE, 1998). This study formed part of a work program to investigate the potential of ethanol blends as alternative motor vehicle fuels, which was developed by the Australian government.

1. APACE Research (1998)

This study is based on the NSW Environmental Protection Authority “Petrohol In-Service Vehicle Emission Study”. The major objective of the study was to determine the effect on exhaust and evaporative emissions of the use of E10 in the existing petrol engine vehicle fleet, with emphasis on:

- regulated emissions of carbon monoxide (CO), oxides of nitrogen (NO_x), and total hydrocarbons (THC);
- Emissions of 1-3 butadiene, benzene, toluene and xylenes (air toxics);
- emission of aldehydes (acetaldehyde, formaldehyde and acrolein);
- OFP; and
- emissions of CO_2.

The study tested 60 in-service light-duty passenger vehicles over a 2-year period from 1995 to 1997. All vehicles were tested according to the ADR37/00, and a total of 188 complete emission tests were conducted in this program. The vehicles were selected from 5 different manufacturers with the major vehicle makes and models (1979 to 1995) being representative of the in-service fleet. The vehicles were all selected from the Newcastle/Central Coast region of NSW and it is was not established if this vehicle sample is representative of the Australia wide in-service fleet. The vehicle sample covered both carburettor and fuel injection type vehicles with a range of odometer readings (7,000 to 440,000 km). The vehicles were grouped into 19 leaded fuelled vehicles (pre-1986 models), and 41 unleaded fuelled vehicles fitted with a catalytic converter (post-1986 models). The post-1986 (includes 1986 year model) model vehicles and contain either a 2-way or 3-way converters. The proportion of 2-way and 3-way catalysts is not known, however, the 3-way Catalytic Converter (TWC) predominate (NGGIC, 1998).

Of the 60 vehicles tested, 37 were tested once (Base Fleet vehicle category) following standard engine tuning. Two additional aims of the study were to determine the effect of maintenance in reducing emissions and determine the deterioration in emissions over a 12-month period. A category of vehicles, Long Time In-Service (LTIS) (11 vehicles), was tested 3 times: post tune (like Base Fleet), and 12 months later pre-tune and post-tune testing.

The ADR37/00 is equivalent to the US Federal Test Procedure (FTP) and may not represent accurately the real on-road driving patterns. The actual on-road emissions are expected to be higher than predicted by the ADR cycle test due to the ADR (and FTP) drive cycles tend to have (Watson, 1995):

- lower acceleration rates and speeds,
- longer idle times; and
- an early and relatively high speed event leading to the catalyst reaching operating temperature relatively quickly.

The regulated exhaust emissions of THC, NO_x and CO, together with CO_2 and fuel consumption, as reported by NSW EPA, are summarised in Table A2. Included in the table are the emissions reported in a national in-service vehicle fleet emission study (FORS, 1996), which shows good general agreement with the results obtained in the NSW EPA study.

Table A2. Summary of exhaust emissions and fuel consumption (NSW EPA Report)

	ADR37/00 – 3 bag results					AS2877			
	No. of tests	THC (g/km)	NO_x (g/km)	CO (g/km)	CO_2 (g/km)	No. of tests	City Fuel (L/100km)	No. of tests	Hwy Fuel (L/100km)
Petrol	59	1.02	1.53	11.71	261.6	59	12.3	54	8.9
E10	59	0.9	1.57	7.98	264.3	59	12.6	54	9.1
Petrol (1986-on)	41	0.66	1.39	8.45	264.4	41	12.1	36	8.8
FORS (1986-on)		*0.54*	*1.15*	*7.83*					
E10 (1986-on)	41	0.57	1.46	6.18	266.2	41	12.5	36	9
Petrol (Pre-1986)	18	1.86	1.84	19.13	255.1	18	12.6	18	9.1
FORS (Pre-1986)		*1.84*	*1.88*	*19.03*					
E10 (Pre-1986)	18	1.66	1.83	12.06	259.9	18	12.7	18	9.3
Petrol (Pre-Tune)	22	1.24	1.68	14.95	254.7	22	12.2	20	8.8
E10 (Pre-Tune)	22	1.15	1.77	12.36	256.4	22	12.5	20	9.1
Petrol (Post-Tune)	22	1.05	1.65	10.13	259.5	22	12	20	8.8
E10 (Post-Tune)	22	0.95	1.72	6.92	261.2	22	12.3	20	9.1
Petrol (LTIS1)	10	0.49	1.3	5.67	264.9	10	11.9	8	9.3
E10 (LTIS1)	10	0.4	1.42	3.93	264	10	12.2	8	9.5
Petrol (LTIS2)	10	0.49	1.43	6.83	266.7	10	12.1	8	9.1
E10 (LTIS2)	10	0.42	1.62	5	268.6	10	12.5	8	9.4

Source APACE 998 Table 5-5:33.

APACE used linear regressions for assessing the exhaust emissions effect of E10. The trend lines indicate that the use of E10 results in a reduction in exhaust THC emission of 13% for "1986-on" (post-1986) and 9% for pre-1986 vehicles (Figure A1).

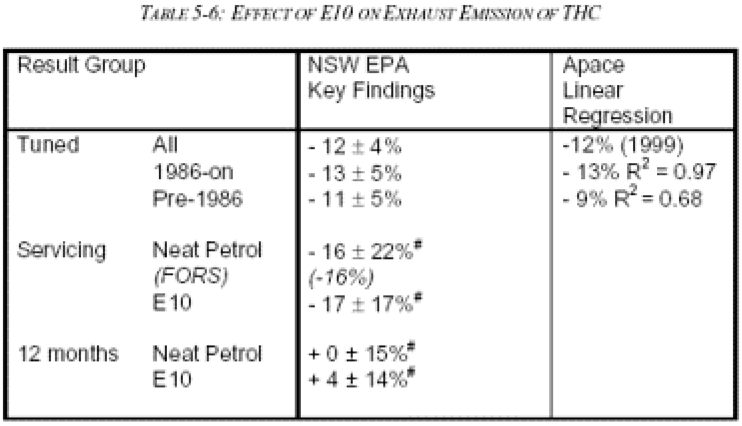

TABLE 5-6: EFFECT OF E10 ON EXHAUST EMISSION OF THC

Result Group		NSW EPA Key Findings	Apace Linear Regression
Tuned	All	- 12 ± 4%	-12% (1999)
	1986-on	- 13 ± 5%	- 13% $R^2 = 0.97$
	Pre-1986	- 11 ± 5%	- 9% $R^2 = 0.68$
Servicing	Neat Petrol	- 16 ± 22%#	
	(FORS)	*(-16%)*	
	E10	- 17 ± 17%#	
12 months	Neat Petrol	+ 0 ± 15%#	
	E10	+ 4 ± 14%#	

Figure A1. E10 impacts on tailpipe HC

Similarly, the use of E10 contributes to a reduction in CO emissions - 30% lower for 1986-on and 38% lower for pre-1986 vehicles (Figure A2).

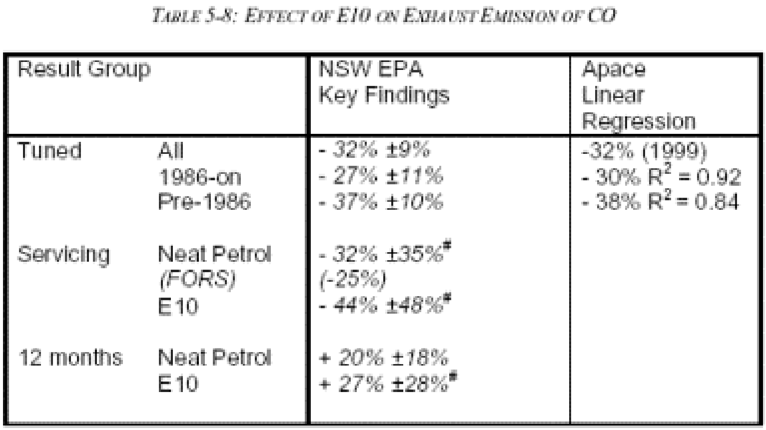

TABLE 5-8: EFFECT OF E10 ON EXHAUST EMISSION OF CO

Result Group		NSW EPA Key Findings	Apace Linear Regression
Tuned	All	*- 32% ±9%*	-32% (1999)
	1986-on	*- 27% ±11%*	- 30% R^2 = 0.92
	Pre-1986	*- 37% ±10%*	- 38% R^2 = 0.84
Servicing	Neat Petrol	*- 32% ±35%*$^{\#}$	
	(FORS)	*(-25%)*	
	E10	*- 44% ±48%*$^{\#}$	
12 months	Neat Petrol	*+ 20% ±18%*	
	E10	*+ 27% ±28%*$^{\#}$	

Figure A2. E10 impacts on tailpipe CO

The use of E10 results in increased NO_x emissions (2%) for 1986-on and decreased NO_x (1%) for pre-1986 vehicles (Figure A3).

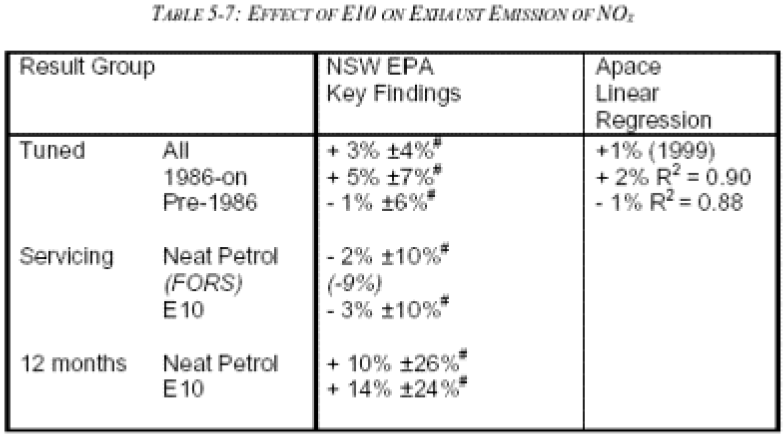

TABLE 5-7: EFFECT OF E10 ON EXHAUST EMISSION OF NO_X

Result Group		NSW EPA Key Findings	Apace Linear Regression
Tuned	All	+ 3% ±4%$^{\#}$	+1% (1999)
	1986-on	+ 5% ±7%$^{\#}$	+ 2% R^2 = 0.90
	Pre-1986	- 1% ±6%$^{\#}$	- 1% R^2 = 0.88
Servicing	Neat Petrol	- 2% ±10%$^{\#}$	
	(FORS)	*(-9%)*	
	E10	- 3% ±10%$^{\#}$	
12 months	Neat Petrol	+ 10% ±26%$^{\#}$	
	E10	+ 14% ±24%$^{\#}$	

Figure A3. E10 impacts on tailpipe NO_x

The increase in exhaust CO_2 when using E10 (Figure A4) is primarily due to the volumetric increase in FC of 3% compared to neat petrol. However, when the full carbon cycle is taken into account it is estimated there is a net reduction of from 5.1 to 7.6% in the mass emission of CO_2 compared to neat petrol. The FORS study (1996) did not report CO_2 emission specifically; however, an overall reduction of 1.5% in fuel consumption was noted following vehicle servicing.

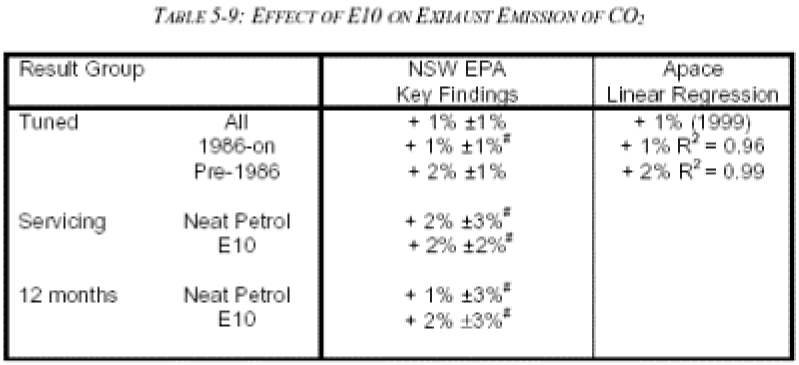

TABLE 5-9: EFFECT OF E10 ON EXHAUST EMISSION OF CO_2

Result Group		NSW EPA Key Findings	Apace Linear Regression
Tuned	All	+ 1% ±1%	+ 1% (1999)
	1986-on	+ 1% ±1%$^{\#}$	+ 1% R^2 = 0.96
	Pre-1986	+ 2% ±1%	+ 2% R^2 = 0.99
Servicing	Neat Petrol	+ 2% ±3%$^{\#}$	
	E10	+ 2% ±2%$^{\#}$	
12 months	Neat Petrol	+ 1% ±3%$^{\#}$	
	E10	+ 2% ±3%$^{\#}$	

Figure A4. E10 impacts on tailpipe CO_2

Aldehyde and air toxics emissions were also collected, during the three phases (Bag 1, 2, 3) of the ADR37/00 emissions tests. The exhaust emissions samples were analysed for approximately 50 species (C_1 to C_{10} hydrocarbons and oxygenated hydrocarbons) by CSIRO–Division of Energy Technology (formerly Division of Coal and Energy. Aldehyde samples were collected from each vehicle but hydrocarbon speciation was conducted only on LTIS vehicles. The study was conducted in two stages: vehicles were tested in the tuned condition with the corresponding conventional petrol as well as with the corresponding petrohol fuel (stage1), and after a period of approximately 12 months the vehicles were retested in both pre-tune and post-tune modes (stage2).

The exhaust emission of air toxics and aldehydes, as reported by NSW EPA, along with additional APACE data set comprised of the emissions from post-tune vehicles are summarised in Table A3.

Table A3. Average mass exhaust emissions of toxics and aldehydes (NSW EPA Report)

		1,3-butadiene (mg/km)	Acetaldehyde (mg/km)	Acrolein (mg/km)	Benzene (mg/km)	Formaldehyde (mg/km)	Toluene (mg/km)	Xylenes (mg/km)
NSW EPA								
Petrol	All	4.8	3.95	1.866	23.54	13.59	36.21	28.94
E10	All	3.79	12.46	2.089	18.16	17.24	28.5	23.11
Petrol	post'86	1.35	2.24	1.138	14.36	5.64	19.21	16.17
E10	post'86	1.23	7	1.227	10.35	7.16	14.17	12.29
Petrol	pre'86	18.6	7.58	3.493	64.83	31.85	112.71	86.43
E10	pre'86	14.02	24.04	4.017	53.3	40.38	93.02	71.77
Petrol	Pre-tune	1.68	4.79	2.417	18.8	16.88	28.83	25.16
E10	Pre-tune	1.38	16.35	2.156	15.66	17.57	24.06	20.4
Petrol	Post-tune	1.77	4.4	2.217	17.87	16.05	26.66	22.2
E10	Post-tune	1.47	13.87	2.118	13.65	18.8	20.47	17.6
Petrol	LTIS1	11.67	1.82	-0.031	30.46	5.52	43.52	36.12
E10	LTIS1	10.1	5.06	0.552	23.74	6.9	34.37	30.39
Petrol	LTIS2	1.77	1.09	0.598	17.87	5.38	26.66	22.2
E10	LTIS2	1.47	4.3	0.865	13.65	6.75	20.47	17.6
APACE								
Petrol	pre'86	28.84	7.55	3.38	88.2	31.12	169.82	140.88
E10	pre'86	24.41	24.2	3.95	65.79	39.27	128.8	105.03
Petrol	post'86	4.19	2.15	1.22	18.08	5.22	22.74	19.7
E10	post'86	3.56	6.3	1.2	13.61	6.56	16.9	15.29

Source APACE 1998 Tables 5-10:36 and 5-11:37.

Simple linear regressions have been fitted for air toxics as well, with the following results:

- increase of formaldehyde with 25% (regardless the age group) –Figure A5;
- increase in acetaldehyde with 171% (for post-1986 group) and with 212% (for pre-1986 group) – Figure A6;
- reduction in acrolein with 5% (for post-1986 group) and increase with 2% (for pre-1986 group) – Figure A7;
- reduction in 1,3-butadiene with 21% (for post-1986 group) and with 14% (for pre-1986 group) – Figure A8;
- reduction in benzene with 26% (for post-1986 group) and with 29% (for pre-1986 group) – Figure A9;
- reduction in toluene with 31% (for post-1986 group) and with 28% (for pre-1986 group) – Figure A10; and
- reduction in xylenes with 27% (for post-1986 group) and with 28% (for pre-1986 group) – Figure A11.

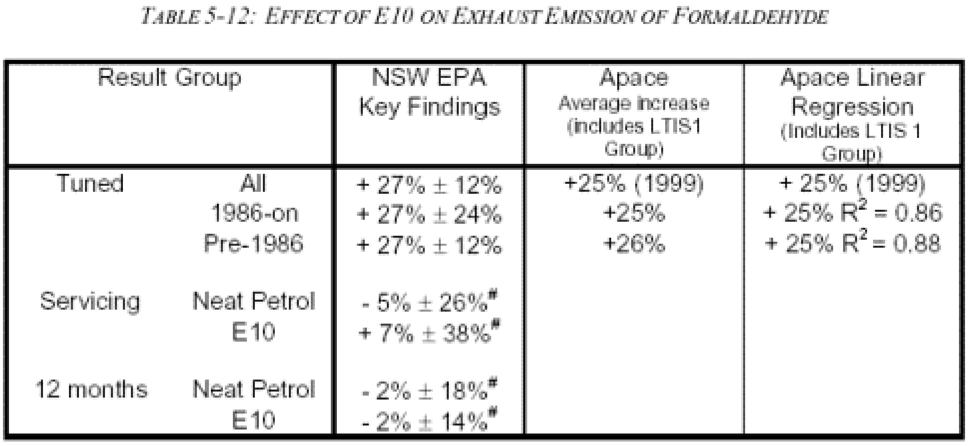
TABLE 5-12: EFFECT OF E10 ON EXHAUST EMISSION OF FORMALDEHYDE

Result Group		NSW EPA Key Findings	Apace Average increase (includes LTIS1 Group)	Apace Linear Regression (Includes LTIS 1 Group)
Tuned	All	+ 27% ± 12%	+25% (1999)	+ 25% (1999)
	1986-on	+ 27% ± 24%	+25%	+ 25% $R^2 = 0.86$
	Pre-1986	+ 27% ± 12%	+26%	+ 25% $R^2 = 0.88$
Servicing	Neat Petrol	- 5% ± 26%#		
	E10	+ 7% ± 38%#		
12 months	Neat Petrol	- 2% ± 18%#		
	E10	- 2% ± 14%#		

Figure A5. E10 impacts on tailpipe formaldehyde

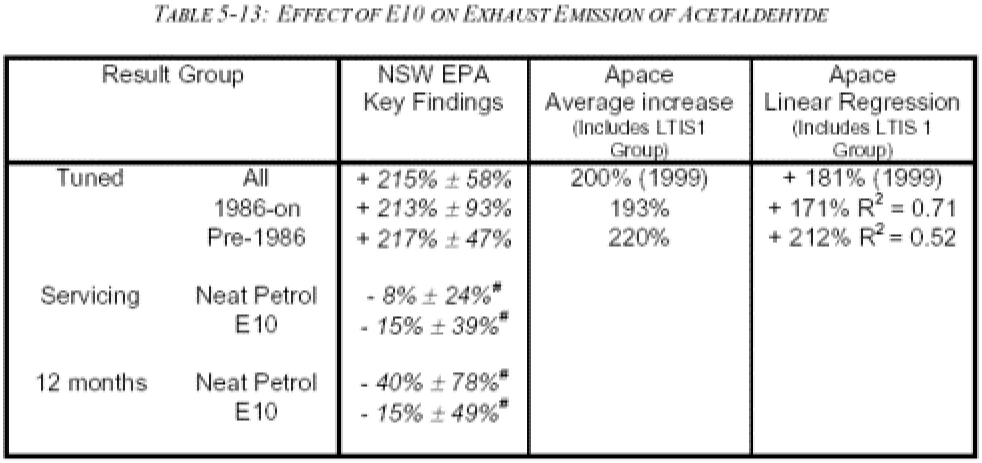
TABLE 5-13: EFFECT OF E10 ON EXHAUST EMISSION OF ACETALDEHYDE

Result Group		NSW EPA Key Findings	Apace Average increase (Includes LTIS1 Group)	Apace Linear Regression (Includes LTIS 1 Group)
Tuned	All	*+ 215% ± 58%*	200% (1999)	+ 181% (1999)
	1986-on	*+ 213% ± 93%*	193%	+ 171% $R^2 = 0.71$
	Pre-1986	*+ 217% ± 47%*	220%	+ 212% $R^2 = 0.52$
Servicing	Neat Petrol	*- 8% ± 24%#*		
	E10	*- 15% ± 39%#*		
12 months	Neat Petrol	*- 40% ± 78%#*		
	E10	*- 15% ± 49%#*		

Figure A6. E10 impacts on tailpipe acetaldehyde

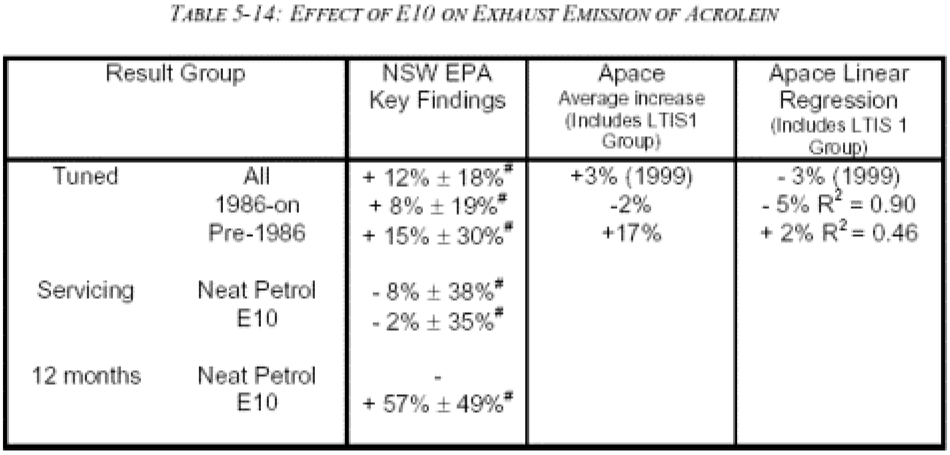
TABLE 5-14: EFFECT OF E10 ON EXHAUST EMISSION OF ACROLEIN

Result Group		NSW EPA Key Findings	Apace Average increase (Includes LTIS1 Group)	Apace Linear Regression (Includes LTIS 1 Group)
Tuned	All	+ 12% ± 18%#	+3% (1999)	- 3% (1999)
	1986-on	+ 8% ± 19%#	-2%	- 5% $R^2 = 0.90$
	Pre-1986	+ 15% ± 30%#	+17%	+ 2% $R^2 = 0.46$
Servicing	Neat Petrol	- 8% ± 38%#		
	E10	- 2% ± 35%#		
12 months	Neat Petrol	-		
	E10	+ 57% ± 49%#		

Figure A7. E10 impacts on tailpipe acrolein

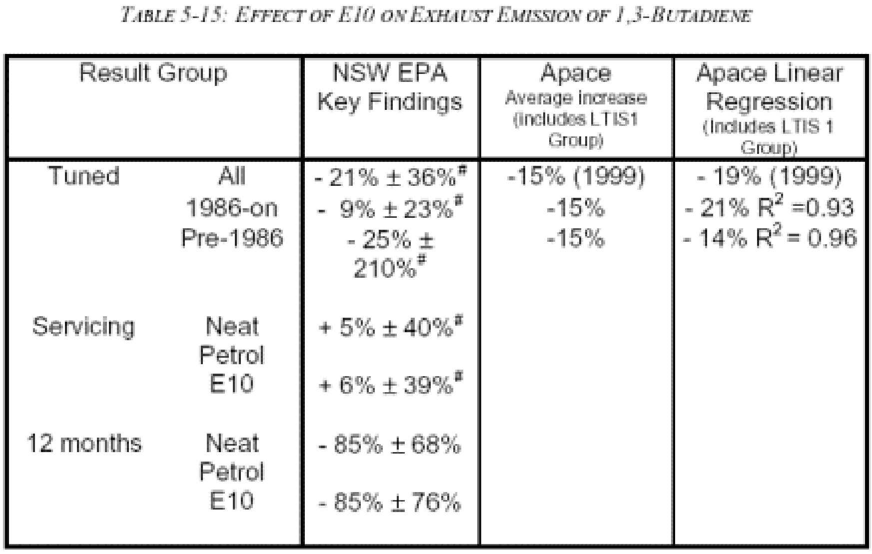

TABLE 5-15: EFFECT OF E10 ON EXHAUST EMISSION OF 1,3-BUTADIENE

Result Group		NSW EPA Key Findings	Apace Average increase (includes LTIS1 Group)	Apace Linear Regression (Includes LTIS 1 Group)
Tuned	All	- 21% ± 36%#	-15% (1999)	- 19% (1999)
	1986-on	- 9% ± 23%#	-15%	- 21% R^2 =0.93
	Pre-1986	- 25% ± 210%#	-15%	- 14% R^2 = 0.96
Servicing	Neat Petrol	+ 5% ± 40%#		
	E10	+ 6% ± 39%#		
12 months	Neat Petrol	- 85% ± 68%		
	E10	- 85% ± 76%		

Figure A8. E10 impacts on tailpipe 1,3-butadiene

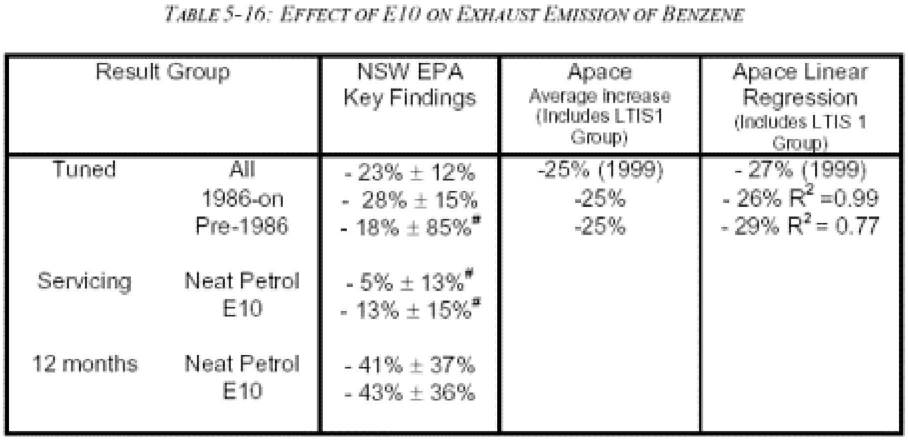

TABLE 5-16: EFFECT OF E10 ON EXHAUST EMISSION OF BENZENE

Result Group		NSW EPA Key Findings	Apace Average increase (Includes LTIS1 Group)	Apace Linear Regression (Includes LTIS 1 Group)
Tuned	All	- 23% ± 12%	-25% (1999)	- 27% (1999)
	1986-on	- 28% ± 15%	-25%	- 26% R^2 =0.99
	Pre-1986	- 18% ± 85%#	-25%	- 29% R^2 = 0.77
Servicing	Neat Petrol	- 5% ± 13%#		
	E10	- 13% ± 15%#		
12 months	Neat Petrol	- 41% ± 37%		
	E10	- 43% ± 36%		

Figure A9. E10 impacts on tailpipe benzene

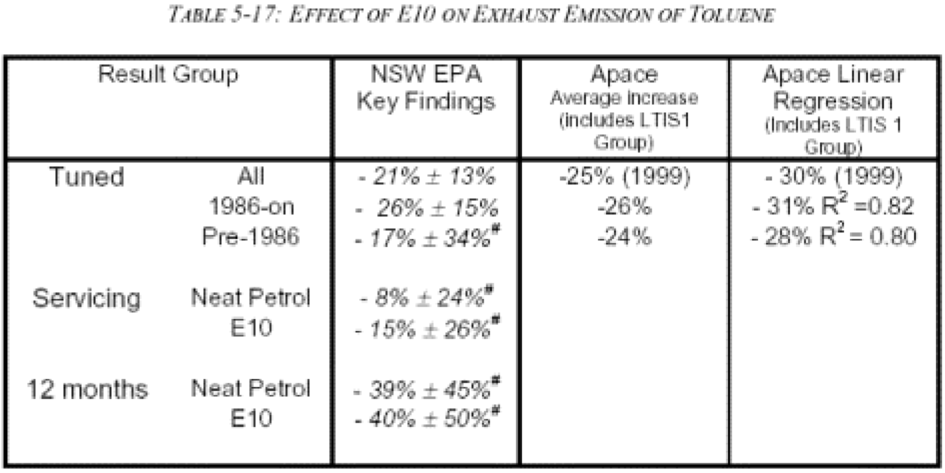

TABLE 5-17: EFFECT OF E10 ON EXHAUST EMISSION OF TOLUENE

Result Group		NSW EPA Key Findings	Apace Average increase (includes LTIS1 Group)	Apace Linear Regression (Includes LTIS 1 Group)
Tuned	All	*- 21% ± 13%*	-25% (1999)	- 30% (1999)
	1986-on	*- 26% ± 15%*	-26%	- 31% R^2 =0.82
	Pre-1986	*- 17% ± 34%#*	-24%	- 28% R^2 = 0.80
Servicing	Neat Petrol	*- 8% ± 24%#*		
	E10	*- 15% ± 26%#*		
12 months	Neat Petrol	*- 39% ± 45%#*		
	E10	*- 40% ± 50%#*		

Figure A10. E10 impacts on tailpipe toluene

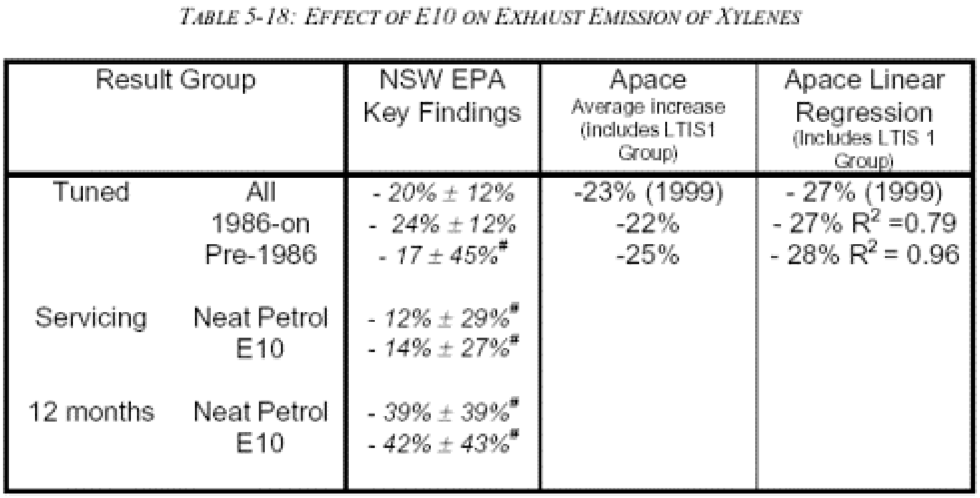

TABLE 5-18: EFFECT OF E10 ON EXHAUST EMISSION OF XYLENES

Result Group		NSW EPA Key Findings	Apace Average increase (Includes LTIS1 Group)	Apace Linear Regression (Includes LTIS 1 Group)
Tuned	All	*- 20% ± 12%*	-23% (1999)	- 27% (1999)
	1986-on	*- 24% ± 12%*	-22%	- 27% R^2 =0.79
	Pre-1986	*- 17 ± 45%#*	-25%	- 28% R^2 = 0.96
Servicing	Neat Petrol	*- 12% ± 29%#*		
	E10	*- 14% ± 27%#*		
12 months	Neat Petrol	*- 39% ± 39%#*		
	E10	*- 42% ± 43%#*		

Figure A11. E10 impacts on tailpipe xylenes

The ozone reactivity or ozone forming potential (OFP) of vehicle exhaust emissions is dependent on the local ambient air characteristics being a complex function of: chemical composition and mass of the emissions; ambient concentrations of pollutants; chemical reaction rates, and meteorology.

Total ozone reactivities (mg O_3/mg NMOG) and total OFP (mg O_3/km) were calculated in the Petrohol study using the Maximum Incremental Reactivity (MIR) method (Carter, 1994) for determining ozone impacts. Incremental reactivities would be expected to give good approximations to effects on ozone with the introduction of petrohol (Chang and Rudy, 1990). The emission reactivities and OFP are presented in Figure A12. There was little variation in reactivities between petrol and petrohol fuelled vehicles for either the pre-or post-1986 vehicles. This resulted in a reactivity adjustment factor (RAF) for petrohol close to 1 for both vehicle groups. For petrohol vehicles lower mass emissions of ozone per km were found. This was due to lower mass emissions of NMOC (approximately 20% lower) and not due to significant reductions in exhaust reactivity.

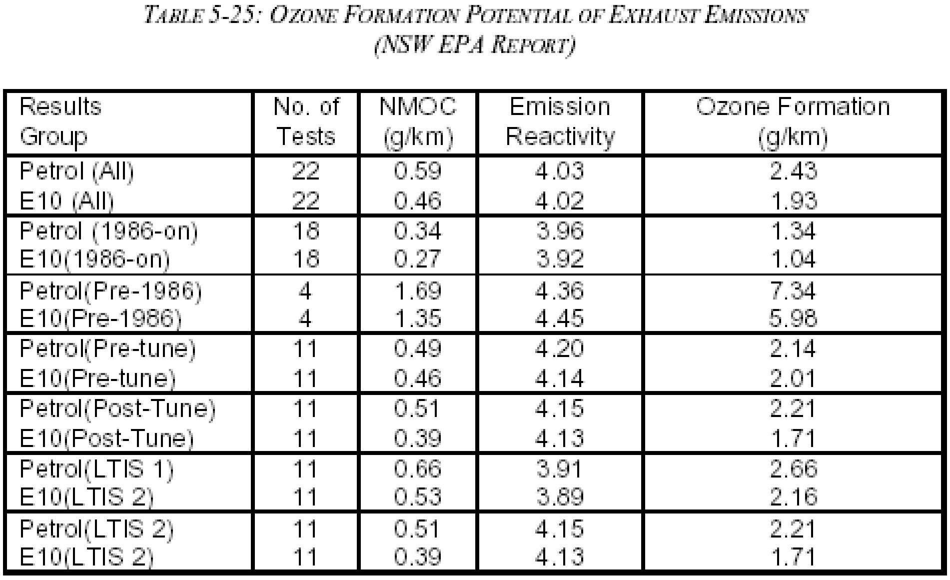

TABLE 5-25: OZONE FORMATION POTENTIAL OF EXHAUST EMISSIONS (NSW EPA REPORT)

Results Group	No. of Tests	NMOC (g/km)	Emission Reactivity	Ozone Formation (g/km)
Petrol (All)	22	0.59	4.03	2.43
E10 (All)	22	0.46	4.02	1.93
Petrol (1986-on)	18	0.34	3.96	1.34
E10(1986-on)	18	0.27	3.92	1.04
Petrol(Pre-1986)	4	1.69	4.36	7.34
E10(Pre-1986)	4	1.35	4.45	5.98
Petrol(Pre-tune)	11	0.49	4.20	2.14
E10(Pre-tune)	11	0.46	4.14	2.01
Petrol(Post-Tune)	11	0.51	4.15	2.21
E10(Post-Tune)	11	0.39	4.13	1.71
Petrol(LTIS 1)	11	0.66	3.91	2.66
E10(LTIS 2)	11	0.53	3.89	2.16
Petrol(LTIS 2)	11	0.51	4.15	2.21
E10(LTIS 2)	11	0.39	4.13	1.71

Figure A12. OFP of exhaust emissions

APACE (1998) found that 10% (v/v) ethanol/petrol blend offers significant benefits in terms of reductions in exhaust and GHG emissions with no apparent detrimental effect on other aspects of engine or vehicle performance.

The 1999 fleet composition[1] has been considered for the project. The use of E10 had the following effects:

- ♦ Regulated exhaust emissions:
  - CO decreases by approximately 32%;
  - THC decreases by approximately 12%; and
  - NO_x increases by approximately 1%.
  - Non-regulated exhaust emissions:
- ♦ Toxics:
  - 1-3 butadiene decreases by approximately 19%;
  - benzene decreases by approximately 27%;
  - toluene decreases by approximately 30%;
  - xylenes decrease by approximately 27%;
- ♦ Aldehydes:
  - formaldehyde increases by approximately 25%;
  - acetaldehyde increases by approximately 180%;
  - acrolein increases by about 5% (indicative only);
- ♦ Carbon Dioxide:
  - exhaust CO_2 increases by 1%; however,
  - net CO_2 emission decreases by up to 7% on full carbon cycle basis;
- ♦ Evaporative emissions (SHED test method - ADR37 protocol):
  - "diurnal" increases by approximately 10%;
  - "hot soak" increases by approximately 40%;
  - with the Multiday Diurnal SHED test method little or no increase in evaporative emissions;
- ♦ Ozone formation potential:
  - of exhaust emissions decreases by approximately 22%;
  - of the evaporative emissions (ADR37 SHED test method) increases by 29%;
  - little or no change in total ozone formation potential after weighting exhaust and evaporative emissions (+0.24%);
- ♦ Health risk assessment of "toxics" and aldehyde emissions:
  - carcinogenic risk decreased by approximately 24% (Environment Defence Fund risk factors);
  - acute and chronic (respiratory, reproductive and neurological) health risks increase by 3% (Environment Defence Fund risk factors);
- ♦ Water tolerance:
  - water content maintained below 1.25% w/w;
  - an ethanol compatible water detecting paste must be used to establish the water content of underground storage tanks;
  - older vehicles are more prone to suffer from phase separation when first fuelled with ethanol/petrol blend, however subsequent continuous use of blend prevents water accumulation within the fuel tank.

There is no discernible effect on any plastic or elastomer materials and no corrosion in fuel wetted metal parts such as fuel tanks, lines, pressure regulators, etc. No additional or unusual wear to that normally expected, and no additional increase in wear metals or decrease in total base number (TBN) of the lubricating oil have been noticed.

The fuel consumption increases by 2.6% for both city and highway cycles.

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[1] Approximately 25% pre-1986 vehicles using leaded petrol (LP) and 75% 1986-on vehicles using unleaded petrol (ULP).

With respect to “hot and cold driveability”, the test results show a reduced tendency for “knock” under both hot and cold conditions.

Further work needs to be undertaken to determine the "real world" evaporative emission from E10. In addition to the environmental benefits, APACE study identifies the following socio-economic benefits:

- increased fuel self-sufficiency with associated improved balance of trade/saved foreign exchange;
- The Australian Bureau of Agricultural and Resource Economics (ABARE) estimates that net annual liquid petroleum imports (including crude oil, LPG, and refined petroleum products) will rise from 3061 ML in 1991-2 to 13,358 ML in 2004-5. In contrast, sufficient ethanol can potentially be produced domestically and renewably from ligno-cellulosic resources to meet all of Australia's liquid fuel demand;
- Expansion of the agricultural economy, value enhancement of existing biomass resources, treatment of land degradation and reforestation;
- Nationwide decentralisation and regional industry development, creating employment in rural areas.

2. Al-Hasan (2003)

The author investigated the effect of ethanol-gasoline blends on engine performance and exhaust emissions on a four stroke, four cylinder SI engine – Toyota Tercel 3A and found that blending increases the torque, brake power (8.3%), volumetric (7%) and brake thermal efficiencies (9%) and fuel consumption (5.7%), while it decreases the brake specific fuel consumption (2.4%) and equivalence air-fuel ratio (3.7%).

The CO and HC emissions decreased (46.5%, and respectively 24.3%), but CO_2 was higher (7.5%). The 20% ethanol blend gave the best results for all measured parameters at all engine speeds and the addition of 25% ethanol to ULP was achieved without any problems during engine operation.

3. Schifter *et al.* (2001)

Schifter *et al*. (2001) conducted vehicle emissions tests (FTP-75) on 12 in-service vehicles (“normal emitters” being representative of the metropolitan area of Mexico city) equipped with a range of pollution control technologies (no catalyst, oxidation catalyst and TWC) using a range of ethanol blended fuels (E3, E6, E10) and were compared to a 5% MTBE petrol. The emission results of the regulated pollutants and air toxics using E10 fuel are given in the following table. A clear trend of decreasing emissions with advancement of pollution control technology is observed. The most unfavourable effect of ethanol addition to the fuel (E3, E6, E10), relative to 5% MTBE petrol, was a considerable increase in acetaldehyde emissions (80-104%) regardless of the pollution control technology. This is supported by other emissions data, showing increased acetaldehyde emissions up to 100% with the use of 2.0 wt.% ethanol oxygenated petrol, which negatively results in the subsequent photochemical production of Peroxy-Acetyl-Nitrates, PAN (Kirchner *et al.,* 1997).

Table A4. Tailpipe emissions of regulated air pollutants (g/km) and air toxics (mg/km) using E10

	No catalyst	Oxidation Catalyst	TWC
Regulated (g/km)			
CO	12.88	8.29	2.97
HC	1.45	0.56	0.22
NO_x	1.75	1.23	0.48
Air Toxics (mg/km)			
Benzene	47.54	28.7	8.11
1,3-Butadiene	18.35	4.5	0.83
Formaldehyde	30.29	2.58	1.01
Acetaldehyde	15.30	7.00	1.62

4. Orbital (2003)

A report to Environment Australia (Orbital, 2003) presented a series of comprehensive vehicle tests, which assessed the impact of petrol containing 20% ethanol on the Australian passenger vehicle fleet. The program is part of the Department of Environment and Heritage project "Market Barriers to the Uptake of Biofuels – Testing Petrol Containing 20% Ethanol (E20)". The vehicle-testing program consisted of 9 vehicles, which were selected to "ensure adequate representation of the Australian passenger vehicle fleet". Although not directly comparable to E10 the study presents valid observations concerning the effect of the addition of ethanol to petrol on: emissions, vehicle performance and driveability, and component material compatibility.

5. Orbital (2002)

The first component of the Department of Environment and Heritage project was a desktop study (Orbital, 2002), which provides a good overview of the effects of higher concentration ethanol fuel blends (>10% v/v). This review concluded that, for vehicle fuel systems with open loop control, CO emission reductions were achieved upon addition of ethanol (5 to 20% ethanol) to the fuel. For HC and NO_x, however, the effect is "dependent on the base vehicle engine calibration and driving conditions". In contrast, vehicles with closed loop control show minimal emission effects (upon ethanol addition) providing "the controller is able to maintain the desired equivalence ratio".

6. Lipman and Delucchi (2002)

Lipman and Delucchi developed emission factor estimates for CH_4 and N_2O from conventional vehicles and estimated relative emissions of CH_4 and N_2O from different alternative fuel vehicles (AFV): passenger cars, light-duty trucks, heavy-duty vehicles.

The authors appreciate that these two GHGs contribute to "*about 15% of the life cycle GHG emissions impact of conventional gasoline vehicles, and up to 43% of the life cycle GHG impact of some alternative fuel vehicle types*". (p.478); For example, CNG (28%), LPG (18%), EtOH90% from wood, grass (43%) – Table 1, p. 479.

The authors also cite the Hansen *et al.* (2000) study that states that an alternative strategy for mitigating potential future climate change should be focusing on non-CO_2 GHGs and black soot aerosol, as they have similar impact to CO_2.

The paper provides a database of these emissions estimates for different vehicle types.

7. NESCAUM (2001)

Ethanol appears to be one of the least toxic of the major components of petrol when considering common toxicological endpoints, such as carcinogenicity and central nervous system depression. Preliminary analyses indicate that direct exposure to fuel ethanol in the air and in contaminated drinking water is not expected to pose public health risks.

The potential for other adverse impacts, including developmental effects, associated with large-scale exposure to low levels of ethanol is uncertain; additional analyses to estimate ambient exposure to ethanol and its atmospheric breakdown products, including highly toxic constituents such as acetaldehyde and PAN are needed to assess the potential public health impacts of increased ethanol use (in Northeast).

Ethanol-blends provide some air quality benefits compared to non-oxygenated blends: lower rates of CO and particulate emissions, as well as GHG benefits. With ethanol, the CO benefits will partially offset the adverse ozone impacts associated with increased NO_x and VOC emissions.

Low-level ethanol contamination of groundwater (i.e., less than 400 µg/L, a draft Water Comparison Value derived in the report) is not expected to substantially alter blood alcohol concentrations or produce a significant health risk.

Due to the rapid biodegradability in the environment, ethanol poses significantly less risk to water resources.

Due to ethanol's affinity for water, infrastructure problems exist with ethanol distribution—ethanol containing petrol cannot be transported through existing pipelines. Ethanol will need to be transported and stored separately from petrol until the point where it is loaded into tanker trucks for delivery to retail stations. Segregated ethanol storage tanks and new blending equipment will be needed at distribution terminals.

8. Poulopoulos *et al.* (2001)

The authors studied the effect of ethanol addition to petrol using a four-cylinder Opel 1.6 L ICE equipped with a hydraulic brake dynamometer, under a variety of engine operating conditions. For exhaust emissions treatment, a typical three-way catalyst was used (Pt/Rh). Three test fuels were used: conventional unleaded petrol, and 3% and 10% ethanol blends (w/w). Relative to the base fuel, addition of 10% ethanol resulted in an increased: oxygen content (from 0.16 to 3.58% w/w); RON (from 95.5 to 98.9); MON (from 86 to 87), and increased RVP, combined with a decreased aromatics content (from 43.7 to 42.9% w/w).

The 10% blend resulted in decreased CO emissions and increased acetaldehyde. Catalytic converter efficiency for acetaldehyde was higher with E10, relative to E3, which could be explained by the enhanced oxygen content, and hence oxidation capability, of the E10.

Aromatic emissions were decreased by ethanol addition to petrol. Ethanol was identified in exhaust gases only on the blends, and in the case of 10% blend was double than for 3% blend.

In addition, methane emissions after the catalytic converter were sometimes higher than after the engine, which confirmed previous work by the authors.

9. Becker *et al.* (2000)

Becker *et al.* (2000) conducted a combined tunnel and dynamometer study (USFTP) to elucidate various N-containing species emissions (including N_2O) from vehicles. The authors established well-correlated emission factors of 4.1 x 10^{-5} and 4.3 x 10^{-5} g N_2O/CO_2 for the tunnel study and dynamometer tests, respectively. The tunnel studies represent vehicles with catalysts operating at full operating temperature. The dynamometer tests were conducted on various passenger cars and trucks (all equipped with TWC) using a variety of fuels and a range of 2–32 mg/km N_2O emissions were found, which was similar to other comparative vehicle dynamometer studies. Although an E10 fuel was not tested in the dynamometer studies, an E85 blend was tested on 1 car and 1 truck and showed low N_2O emissions.

10. He *et al.* (2003)

The authors simultaneously conducted engine out and tailpipe emissions tests using an engine dynamometer and compared the effect of the addition of 10 and 30% (v/v) ethanol to conventional fuel relative to the base fuel. A typical Pt/Rh catalyst was used. It was found that the addition of ethanol on emissions was small relative to the impact of the catalyst. The tailpipe emissions were found to be closely dependent on:

- engine out emissions;
- catalytic converter efficiency;
- engine speed and load; and
- AFR.

The engine out emissions of HC were decreased significantly under all engine operating conditions, while CO and NO_x emissions were not decreased under all conditions.

CONCLUSIONS - TAILPIPE EMISSIONS

Due to the many complex variables influencing motor vehicle emissions, test results indicate both high variances (within vehicle and between vehicles) for the same test condition, as well as different tests and fuels. Large amounts of data are, therefore, required to develop reliable emission rate models that can identify the changes due to alternative fuels. Catalytic converters variables (type, condition, age) are very important for tailpipe emissions, potentially exerting a greater single influence than fuel type.

The vast majority of vehicle emission data currently available for model development are collected through laboratory testing programs, where vehicles are tested on dynamometer driving cycles designed to reflect on-road activities. Insufficient data are available to accurately reflect the on-road emission rates from all types of vehicles (new models and older vehicles that have continued to age). Furthermore, concerns regarding the applicability of laboratory data to reflect on-road emissions are significant. New modal emission rate models (such as CUEDC), developed from second-by-second data collected in laboratories, are predicting emissions as a function of vehicle operating modes. Hard accelerations, high speeds, engine starts, and other modal activities that affect vehicle load affect significantly the emission pollutants (VOCs, CO, and NO_x).

CO emissions tend to move with VOC tailpipe emissions over many operating conditions; that is, both VOC and CO emissions are high when engines are cold and speeds are low.

NO_x emissions vary with the AFR and heat of the engine (exponential with increase in combustion temperature).

The extent to which the vehicle samples selected for testing are representative of the current fleet is an important issue, such that quantifying their emissions may not bias the emissions modelling. From the test data provided, it appears that some vehicles are high "emitters" (exhibit high emission rates under many operating conditions).

The majority of tailpipe emission studies are restricted to the regulated pollutants (CO, NO_x, HC) and limited data is available concerning GHG emissions as well as PM and air toxics. The data is even more restricted with respect to the effects of biodiesel and E10. The primary source of emissions data for E10, for the purposes of the present study, is the NSW EPA Petrohol study. Non-CO_2 GHG emission factors were not determined in this study, however, reasonable estimates of CH_4 emissions can be derived from the CSIRO HC speciated study (component of the Petrohol study). Based on very limited data (and the primary role of TWC in N_2O production), the N_2O emission factors would not be expected to be significantly affected by E10 (or biodiesel).

Evaporative emissions are a key issue in the use of ethanol blended fuels—the addition of 10% (v/v) ethanol to conventional fuel increases the volatility characteristics of the blended fuel (RVP increased by ~1 psi/7 kPa). The quantification of evaporative emissions is difficult with existing motor vehicle emission legislation but could potentially contribute a significant proportion of the total vehicle HC emissions from vehicles using E10 (evaporative emissions from of biodiesel would not be expected to be significance).

Fuels containing ethanol would be expected to increase evaporative emissions due to:

- higher resultant vapour pressures leading to greater HC losses;
- increased ethanol permeability characteristics of rubber, elastomers and other material; and
- ethanol may reduce the working capacity of HC canisters (charcoal);
- evaporative emissions need to be evaluated related to Australian conditions, including emissions performance of the Australian fleet and current refinery practice.

APPENDIX II. LITERATURE REVIEW – BIOFUELS LIFE CYCLE ANALYSIS

INTRODUCTION

The scope of this review is to examine existing local and international literature, studies and scientific reports to assess the net environmental (greenhouse, air quality and other) of replacing fossil fuels with biofuels, produced in Australia from renewable resources, in the Australian transport mix. Where possible, studies applicable to Australian conditions were targeted. The examination of the emissions associated with agricultural production of crops and fuel production where restricted to accounting in the life cycle analyses and environmental issues specific to these processes were not examined in this review (e.g. land/water degradation due to agricultural production). Where possible, original publications cited in references were sourced.

The biofuels considered were biodiesel for heavy duty vehicles (as 100% or 20% and 5% blend with diesel) and 10% (v/v) ethanol in conventional petrol (E10 or "petrohol") for passenger vehicles. The biofuel or ethanol can be sourced from a variety of crops.

The review is grouped into "Pre-combustion and Life cycle Analysis" studies and "Combustion" studies focussing specifically on tailpipe emissions.

Pre-combustion and LCA

Life cycle analyses (LCA) are necessary to assess the overall GHG and air pollutant emissions implications for individual biofuel applications because the GHG abatement of biofuels emissions is strongly dependent on the feedstock and the production processes used. These pre-combustion processes (or "upstream" processes) need to be fully characterised to more accurately assess the LCA emissions. The objective is to determine the total inputs and outputs related to the production and combustion of the fuel. The modelling of these processes and associated variables requires numerous assumptions rendering life cycle assessments of alternative fuels and is a complex issue.

A LCA can be conducted according to various methodologies, which are governed by an international standard: ISO 14040 series (International Standards Organisation, 1998). Key analysis issues are system boundaries used and allocation of emissions for co-products and waste-products.

LCA allows the direct comparison between different fuels, resulting in a full-fuel cycle (or "well to wheel") analysis of greenhouse gas emissions for each fuel. This allows the emissions common to all fuels, those associated with vehicle manufacture and disposal, transport infrastructure, etc to be removed from the analysis.

Vehicle emissions are usually expressed in terms of mass of pollutant per km (g/km). On this basis, a large or heavy-duty vehicle will have significantly higher emissions than a small or light-duty vehicle. To overcome the impact of vehicle size on emission rates, the emissions can be normalised to power output or unit fuel consumption by using a "standard" vehicle. When full fuel-cycle emissions are evaluated; however, the split between non-vehicle and vehicle emissions varies according to vehicle classification.

The LCA studies examined in this review were prioritised according to the following criteria:
- Applicability to Australian conditions;
- Current relevance;
- Detailed process information using actual data, and
- LCA conducted using accepted methodologies.

Combustion

Fuel combustion in motor vehicles is a highly complex process, with large numbers of variables influencing tailpipe emissions. The review of studies investigating tailpipe emissions from the use of biofuels is discussed in Appendix I.

BIOFUELS LIFE CYCLE ANALYSIS (LCA)

The most extensive life cycle analysis (LCA) of biodiesel and E10, as well as other alternative fuels, in the Australian context has been conducted by CSIRO (Beer *et al.,* 2000 and 2001). These studies consisted of literature reviews and desktop analyses assessing the environmental and health-related issues, viability and functionality of alternative fuels. Within these studies are additional comprehensive literature reviews of previous related studies. Three classes of emissions were considered:

- greenhouse gases (CO_2, N_2O, CH_4, hydrofluorocarbons, sulfur hexafluoride, and perfluorocarbons),
- air pollutants (CO, NO_x, PM, NMHC, sulfur dioxide); and
- air toxics (benzene, toluene, xylene, formaldehyde, acetaldehyde, 1,3 butadiene and PAH).

A number of upstream fuel production process were characterised by generic operating conditions not specific to individual producers and international data was used in the absence of data for Australian conditions. Conditions peculiar to Australia, in particular, require actual data. Real-world process data was frequently difficult to obtain and is common to many studies. Due to the release of more recent or relevant data, the data used for some downstream and upstream processes are no longer valid.

E10 LCA

A number of fuel ethanol life cycle studies have been conducted internationally using various crop sources. The majority of fuel-grade ethanol is produced in Brazil and the USA, with minor production facilities in Europe (France), Canada and Asia (China and India). Almost universally, full fuel life cycle studies show that the production of ethanol is more energy intensive than that of fossil fuel based petrol, which is due to:

- crop production;
- processing;
- transport; and
- distillation.

The CSIRO study (Beer *et al.,* 2001) presented results of E10 as ethanol blended with PULP (PULP was used as the reference fuel for light-duty vehicles) instead of ULP, which is the predominant E10 product in Australia. The exbodied GHG emissions results associated with E10, presented in the CSIRO study, showed increased emissions relative to PULP of 0.1-2.8% from molasses and wheat feedstocks with only wood waste providing the only emission benefit (1.6%).

The negative LCA emissions reported by CSIRO are noteworthy because they are in contrast to many local and international studies discussed below, which show positive emissions benefits from the use of E10.

For Australian ethanol production (Manildra), Saddler (1996) reported full fuel-cycle GHG emissions from ethanol production in the range 39 to 64% relative to petrol production, which is considerably lower than that extracted from the CSIRO study (98 to 103% relative to PULP).

Recently, Energy Strategies in conjunction with APACE Research conducted a LCA on energy use and GHG emissions for the existing Sarina distillery and planned Burdekin distillery for CSR Sugar

(Energy Strategies, 2003). For both distilleries, the life cycle GHG emissions for E10 were calculated using vehicle emission results from the Petrohol study (APACE, 1998) and compared to the emissions from conventional petrol (upstream emissions calculations followed AGO recommendations, (AGO, 2003). Using the conventional (economic) allocation, the life cycle comparison of E10 with petrol for the Sarina and Burdekin distilleries was a reduction of 3.1 and 7.1% respectively. Using the expanded system boundary method, the reductions were 4.6 and 3.9% for the respective distilleries. These estimates can be compared to the CSIRO findings for E10 produced from C molasses (compared to PULP) of 2.8 and 0.7% increases using the allocation and expanded system boundary methods, respectively (Beer *et al*., 2001) Energy Strategies compared the analyses conducted by CSIRO to theirs (Energy Strategies, 2002 and 2003) and stated the following major differences in data, assumption and method followed related to the Sarina distillery:

- actual and more up-to-date operation data used by Energy Strategies;
- Sarina operations use a bagasse/black coal energy split of 70/30 (CSIRO assumed a 50/50 split but also applied a 100% black coal energy source using the economic allocation analysis);
- CSIRO calculations were based on E10 blended with PULP and not ULP;
- CSIRO E10 tailpipe emissions data not valid (note: this was due to a lack of suitable data available at the time);
- CSIRO did not include allocation of emissions to the production of bio-dunder;
- actual production and market prices of C molasses and ethanol used in the allocation method used by Energy Strategies;
- incomplete expanded system boundary method analysis conducted by CSIRO with respect to omission of energy and emissions of bio-dunder and the alternative uses of molasses; and
- CSIRO did not include the emissions and energy associated with the avoidance of production of equivalent amount of petroleum fuels.

There are, however, other studies that show similar results (possible negative emissions benefits) to the CSIRO analysis. These include IPCC (1995), IEA (1999) and Pimentel (2003). The IPCC report, based on results from Delucchi (1993), estimated life cycle GHG emissions (g/km) for light-duty vehicles using ethanol prepared from sugar cane, corn and wood and compared them to petrol. The emissions associated with the supply of the differently sourced fuels impacted significantly on the total emissions with corn produced ethanol having the highest range of total emissions and having potentially higher emissions than petrol. This was supported by Pimentel (2003) who stated that for ethanol fuels derived from US corn production, the environmental impacts and energy balance (as well economics) are negative. Pimentel has frequently commented that the promise of ethanol as a fuel is overstated as it is uneconomical to produce. There is considerable debate in the US about the validity of ethanol as an alternative fuel. Graboski, a consultant for the National Corn Growers Association in the US, reasons that Pimentel's findings are based on out-of-date farming and ethanol processing data and that recent improvements in agricultural efficiency and ethanol processing plants have made corn-to-ethanol production much more cost effective (http://www.ncga.com/public_policy/issues/2001/ethanol/08_22_01b.htm).

The opposing findings (of net emissions effects) of these studies could be attributed to a number of factors associated with upstream emission calculations:

- system boundaries used (what is and what is not included in the LCA);
- allocation of emissions for co-products and waste products;
- transport and supply of feedstocks;
- efficiencies of farming practices, and
- processing plant efficiencies.

Examples of system boundaries considered are: cogeneration (utilisation of energy waste); fuel substitution, and inclusion or exclusion of other miscellaneous processes. An example of a unique characteristic of E10 production in Australia is that to produce ethanol, CSR bulk ship from the site of production in Queensland to Melbourne (Yarraville) to produce anhydrous ethanol (the quality required for blending with petrol) after which the product is returned to Queensland for blending and distribution to fuel stations-incurring considerable emission losses associated with product handling and shipping (Energy Strategies, 2003). Current analyses would incorporate cogeneration using bagasse (waste sugar product) and less energy intensive dehydration processing replacing distillation with molecular sieves.

In addition, downstream emission factors are also potential sources of differences between studies. The CSIRO study used the best available data available at the time, including tailpipe emission data presented by MacLean (1998) in a PhD dissertation – the data of which was not obtained from actual emissions testing of E10 fuel nor was particularly applicable to Australian conditions. This data has been superceded by recent Australian tailpipe emission results from E10 fuelled vehicles collected by NSW EPA for APACE Research (APACE, 1998).

The CSIRO study (2001) assumed that 1 MJ of fuel was consumed in both the petrol and the E10 examples. However, on calculating the carbon content of the 90% petrol and adding this to the non-fossil carbon, the total value is 4 grams short of the observed emission. Clearly the E10 vehicle is using more fuel to provide the same drive to the vehicle. The result of this recalculation is shown in Table A5. The net effect of this recalculation is that approximately 6% more ethanol is required; however the total fossil CO_2 emission goes from 70.1 g per MJ of fuel down to 69.9 g for 1.05 MJ of E10 required to deliver the same end use energy (and thus the same MJ) of petrol fuel.

On the basis of this recalculation the use of E10 confers a slight (1.7%) tailpipe greenhouse gas emissions benefit compared to petrol. Whether the use of E10 confers a greenhouse gas benefit or a greenhouse gas penalty depends, to a first approximation, on whether the resulting drop in fuel economy is more, or less, than 10%. The situation is complicated by the fact that the evaporative emissions of the ethanol-petrol azeotrope are greater than the evaporative emissions of either petrol or ethanol on its own. Thus in-service performance will show a lower efficiency than theoretical calculations or laboratory tests as a result of the increased loss of fuel through evaporation.

Apace Research Ltd (1998) found that, in their field trials of E10 (based on 10% ethanol by volume), there was a 1% loss in efficiency (expressed as a 1% increase in CO_2) but a 7% decrease in fossil CO_2. Beer *et al.* (2001) claimed that tailpipe emissions of E10 have a greenhouse gas penalty. Our re-calculation assigns a slight tailpipe greenhouse gas benefit.

Table A5. Calculation of fossil and non-fossil CO_2 from PULP E10 on the basis of 1 MJ work

	Estimated g CO_2 per MJ	Vehicle Energy MJ/km	Energy content of fuel (MJ/kg)	Fuel g	PULP in E10 g	CO_2 per g fuel	Fossil CO_2 per MJ fuel	Non-fossil CO_2 per MJ fuel
PULP	71.07	2.42	43.1	23.20		3.06		
PULP E10	70.66		41.7	23.97	21.58		66.09	4.57
PULP E10 delivering same work as 1 MJ PULP	74.71	2.56	41.7	25.35	22.81		69.88	4.83

E10 Pre-combustion (Upstream)

The main crops in Australia currently used for ethanol production are molasses (by product of sugar production) and wheat. Potential crops are sugar cane and sorghum as well as a variety of others including sugar beet, crop residues and timber. In Australia, the key production processes of ethanol fuel derived from molasses are considered to be based on mature technology (low grade “C molasses” is the primary feedstock). Future efficiencies gains are possible, with the most potential from greater utilisation of waste products and co-products. Ligno-cellulosic based ethanol production is, however, at a considerably less mature stage of development with large potential gains in efficiencies. There are a variety of potential ligno-cellulosic sources, one of which is bagasse (sugar cane residue). Currently bagasse is used as a cogeneration fuel in the sugar processing plants and as a source of heat for molasses-based ethanol production (mainly for distillation). The major waste product of sugar cane processing is "dunder", which has potential uses as a fertiliser or could undergo methanation to produce co-generation fuel for plant operation.

In the future it would be expected that ethanol production costs would decrease with advancements in biotechnology (Lugar and Woolse, 1999).

Biodiesel LCA

The majority of biodiesel is produced in Europe (RME) and USA (soy-based). Austria was one of the first main transport users of biodiesel.

A number of biodiesel life cycle studies have been conducted internationally using various crop sources. These have been reviewed previously by CSIRO (Beer *et al.,* 2000 and 2001). CSIRO conducted the most extensive series of biodiesel LCA, for Australian conditions. The first stage of the comparison of transport fuels study by Beer *et al.* (2000) used the tailpipe emissions from a US bus study (Sheehan *et al.*, 1998a) with the distribution of life cycle emissions studies by Sheehan *et al.* (1998a,b) to estimate fuel life cycle emissions for BD20 (Table A6) and BD100 (Table A7).

Table A6. Fuel life cycle emissions (g/km) of a bus using 20% biodiesel

Emission Class	Pre-combustion	Combustion	Total
CO_2	37.7	1,312	1,350
CH_4	0.48	-	0.48
N_20	0.08	0.02	0.10
CO	3.01	4.28	7.29
NO_x	1.19	23.51	24.70
NMVOC	1.64	-	1.64
Particles	0.250	0.378	0.628

Table A7. Fuel life cycle emissions (g/km) of a bus using 100% biodiesel

Emission Class	Pre-combustion	Combustion	Total
CO_2	512	1,189	1,701
CH_4	0.219	0.018	0.237
N_20	0.443	0.025	0.468
CO	3.76	6.25	10.28
NO_x	3.02	10.51	13.53
NMVOC	2.16	0.79	2.95
Particles	1.15	0.5	1.65

Beer *et al.* (2001) used data from the Apelbaum Consulting Group (1997) for the passenger task and the freight task in Australia and took the mean energy intensity for the Australian freight task to be 1.2 MJ/tonne-km (Apelbaum Consulting Group, 1997: p.118), and the energy intensity of buses to be 1.06 MJ/passenger-km (Apelbaum Consulting Group, 1997: p.116). The comparative life cycle emissions for buses are reproduced in Table A8.

Table A8. Urban and total life cycle emissions per km for buses calculated for diesel and biodiesel

Full LC		LS Diesel	Canola biodiesel	Soybean biodiesel	Rape biodiesel	Tallow biodiesel	Tallow alternative allocation	Waste cooking oil biodiesel	Waste cooking oil alternative allocation
Greenhouse	kg CO_2	1.66	0.77	0.58	0.79	0.75	0.89	0.13	0.13
NMHC total	g HC	2.71	2.58	3.07	2.60	2.53	1.08	1.07	1.09
NMHC urban	g HC	2.14	2.39	2.91	2.39	2.33	1.06	1.05	1.07
NO_x total	g NO_x	20.20	23.15	22.94	23.48	23.08	21.16	21.12	21.21
NO_x urban	g NO_x	19.10	21.77	22.07	21.81	21.75	21.14	21.11	21.20
CO total	g CO	4.89	3.05	3.92	3.06	3.03	2.53	2.52	2.60
CO urban	g CO	4.69	2.77	3.75	2.78	2.76	2.52	2.51	2.60
PM10 total	mg PM10	787	534	525	545	533	493	493	493
PM10 urban	mg PM10	760	507	508	508	507	493	492	492
Energy Embodied	MJ LHV	22.8	7.4	8.1	7.6	7.3	3.0	2.9	3.0

The aim of this review is to include the most recent data and determine the relevance to and impact on the previous CSIRO reviews.

The main benefit of biodiesel is the reduced GHG emissions over the life cycle of the fuel. The studies have shown that biodiesel life cycle emissions are lower due to the lower fossil fuel energy demand required in the production of biodiesel, relative to petroleum based diesel. There are, however, more significant N_2O emissions effects associated with agricultural practises used in crop production. The value of by-products formed from biodiesel production is also an issue in conducting LCA.

According to US results, using BD100 (soybean feedstock) in urban buses, relative to 500 ppm LSD substantially reduces life cycle emissions of total PM (32%), CO (35%) and SO_x (8%) (Sheehan, 1998). BD100 increased LCA NO_x emissions by 13%, primarily due to increased tailpipe emissions. LCA HC emissions for BD100 were 35% higher with most of this increase due to soybean farming and production, while tailpipe HC are 37% lower than diesel.

The British Association for Bio Fuels and Oils (BABFO) summarised the life cycle emissions from diesel and biodiesel for the UK (EcoTec Research and Consulting Ltd, 1999). The main effect was larger full fuel-cycle emissions of PM from biodiesel when compared to diesel, although this may be different when compared with LSD that generally has lower PM emissions.

Biodiesel Pre-combustion (Upstream)

The production of biodiesel fuel requires less fossil fuel energy than that required for diesel. The GHG emissions arising from the trans-esterification process depend primarily on the amount of fossil fuel involved in the production of the alcohol as given by Sheehan *et al.* (1998a, b). For example, if methanol is used, overall emissions will be higher because current production of methanol involves solely fossil-fuel feedstocks (natural gas or coal). By contrast, if the use of ethanol produced from renewable resources (biomass) using bioprocesses is utilised, GHG emissions will be lower. Methanol can be produced by the gasification of biomass but this is currently not done.

From the point of view of the trans-esterification process itself, differences in molecular structure of the different vegetable oils is insignificant in terms of process parameters or energy demand.

Another source of differences in life cycle emissions of biodiesel arises at the stage of oil and tallow production. In the case of oil-seed crops, there needs to be accounting for energy and raw materials inputs into fertiliser production, land cultivation, materials transportation, harvesting and oil extraction. Similarly, when tallow is used as a feedstock, energy expended in farming activities needs to be accounted for. In both cases appropriate allocation procedures for multiple product streams need to be observed.

CONCLUSIONS - LCA EMISSIONS

Vehicle biofuels, derived from renewable sources, have low GHG emissions (for biodiesel about 65%); biodiesel produces negative GHG during fuel supply (WTT), because the C is removed from the atmosphere during the growth of the plants and will be emitted during combustion (TTW). GHG emission reductions are significantly lower for E10 as would be expected due to 90% of the fuel originating from fossil fuel sources. The crop source used for ethanol production, the upstream emissions associated with the crop production and subsequent ethanol production, are key factors in determining the LCA GHG emissions. A major energy intensive step in the production of fuel ethanol is distillation.

The use of biodiesel in conventional CI engines can result in substantial emissions reductions of HC, CO, and PM. Emissions of NO_x and ultra-fine PM are not significantly changed.

The overall fuel life cycle GHG balance for RME blends offers minor reductions of GHG in comparison to fossil-based fuels.

LITERATURE REVIEWED

Beer *et al.,* 2001

This extensive study assessed the environmental and health-related issues, viability and functionality of 14 fuels used for heavy vehicles (HV) in Australia. The fuels are compared on the basis of life cycle emissions per energy unit and per km. The study used SimaPro5 software and provided confidence intervals for the estimates.

Some of the findings, related to the current study, are summarised in the following:

a. all forms of biodiesel are more friendly to the environment than diesel and have no health risks; biodiesel from vegetable oil is comparable to diesel in its exbodied emissions, exception NO_x and PM;
b. the use of biodiesel changes significantly the characteristics of the heavier HC species as compared to diesel;
c. biodiesel in blends of 20-30% with diesel is expected to comply with all Euro4 standards for HV;
d. oxygenated fuels produce less PM due to a more complete combustion;
e. ultra low sulfur diesel (ULSD) produces more exbodied GHG than LSD due to extra processing energy;
f. tailpipe GHG emissions from petrohol (from renewable sources) are lower than from petrol but is offset by reduced fuel economy;
g. source of ethanol is crucial in determining GHG benefit on a life cycle basis;
h. only petrohol made from wood waste has lower exbodied GHG emissions than PULP; and
i. increased evaporative emissions from petrohol indicate the possibility of increased OFP.

The results of the full life cycle (WTW) emissions calculated per MJ and km - along with some default emission factor values – for diesel and biodiesel are presented in the following tables.

Diesel

Table A9. Emission factors for diesel vehicles (g/km)

Vehicle	Light trucks	Medium trucks	Heavy trucks	Buses
CH_4	0.01	0.02	0.07	0.03
N_2O	0.014	0.017	0.025	0.025
NO_x	1.18	3.1	15.29	4.9
CO	1.11	1.82	7.86	2.88
NMVOC	0.53	0.99	3.78	1.56

Source: National Greenhouse Gas Inventory Committee (1998)

Table A10. Total life cycle emissions (per MJ), calculated for LSD (separate for upstream and tailpipe emissions)

	Units	WTW	WTT	TTW	Uncertainty (%)
Greenhouse	kg CO_2	0.0858	0.0191	0.067	10
NMHC total	g HC	0.140	0.0565	0.084	34
NMHC urban	g HC	0.111	0.027	0.084	34
NO_x total	g NO_x	1.044	0.100	0.944	29
NO_x urban	g NOx	0.987	0.043	0.944	29
CO total	g CO	0.253	0.023	0.230	111
CO urban	g CO	0.242	0.012	0.230	111
PM10 total	mg PM10	40.7	5.42	35.26	45
PM10 urban	mg PM10	39.3	4	35.26	45
Energy Embodied	MJ LHV	1.18	1.18	0	

Source: Beer *et al* 2001 Tables 1.18 to 1.20, pp.98-99; Table 1.27, p.101.

Table A11. Total life cycle emissions (per km) calculated for trucks fuelled with LSD (separate for upstream and tailpipe emissions)

	Units	WTW	WTT	TTW	Uncertainty (%)
Greenhouse	kg CO_2	0.9250	0.2060	0.719	9
NMHC total	g HC	1.509	0.609	0.900	50
NMHC urban	g HC	1.192	0.292	0.900	50
NO_x total	g NO_x	11.250	1.080	10.170	30
NO_x urban	g NO_x	10.638	0.468	10.170	30
CO total	g CO	2.723	0.243	2.480	144
CO urban	g CO	2.612	0.132	2.480	144
PM10 total	mg PM10	438.4	58.4	380.00	39
PM10 urban	mg PM10	423.1	43.1	380.00	39
Energy Embodied	MJ LHV	12.70	12.7	0	

Source: Tables 1.21 to 1.23, pp.99-100; Table 1.27, p.101.

Table A12. Total life cycle emissions (per km) calculated for buses fuelled with LSD (separate for upstream and tailpipe emissions)

Full LC	Units	WTW	WTT	TTW
Greenhouse	kg CO_2	1.66	0.37	1.2910
NMHC total	g HC	2.71	1.09	1.616
NMHC urban	g HC	2.14	0.52	1.616
NO_x total	g NO_x	20.20	1.94	18.270
NO_x urban	g NO_x	19.10	0.84	18.270
CO total	g CO	4.89	0.44	4.453
CO urban	g CO	4.69	0.24	4.453
PM10 total	mg PM10	787	104.9	682.3
PM10 urban	mg PM10	760	77.4	682.3
Energy Embodied	MJ LHV	22.8	22.8	0.00

Source: Tables 1.24 to 1.26, pp.100-101.

Ultra Low Sulfur Diesel

Table A13. Total life cycle emissions (per MJ) calculated for ULSD (separate for upstream and tailpipe emissions)

	Units	WTW		WTT		TTW	
		ULSD	ULSD (100% hydro-processing)	ULSD	ULSD (100% hydro-processing)	ULSD	ULSD (100% hydro-processing)
Greenhouse	kg CO_2	0.0881	*0.0877*	0.0222	*0.0218*	0.067	*0.066*
NMHC total	g HC	0.128	*0.131*	0.0614	*0.0642*	0.084	*0.067*
NMHC urban	g HC	0.097	*0.098*	0.030	*0.031*	0.084	*0.067*
NO_x total	g NO_x	0.915	*0.909*	0.120	*0.114*	0.944	*0.795*
NO_x urban	g NO_x	0.855	*0.844*	0.060	*0.049*	0.944	*0.795*
CO total	g CO	0.314	*0.313*	0.027	*0.026*	0.230	*0.287*
CO urban	g CO	0.303	*0.301*	0.016	*0.014*	0.230	*0.287*
PM10 total	mg PM10	31.9	*32.2*	5.84	*6.16*	35.26	*26.08*
PM10 urban	mg PM10	30.4	*30.6*	4.33	*4.55*	35.26	*26.08*
Energy Embodied	MJ LHV	1.27	*1.34*	1.27	*1.34*	0	*0*

Source: Tables 2.14 to 2.16, pp.115-116.

Table A14. Total life cycle emissions (per km) calculated for ULSD (separate for upstream and tailpipe emissions)

	Units	WTW		WTT		TTW	
		ULSD	ULSD (100% hydro-processing)	ULSD	ULSD (100% hydro-processing)	ULSD	ULSD (100% hydro-processing)
Greenhouse	kg CO$_2$	0.9470	*0.9270*	0.2290	*0.2090*	0.718	*0.718*
NMHC total	g HC	1.363	*1.346*	0.633	*0.616*	0.730	*0.730*
NMHC urban	g HC	1.036	*1.026*	0.306	*0.296*	0.730	*0.730*
NO$_x$ total	g NO$_x$	9.900	*9.750*	1.240	*1.090*	8.660	*8.660*
NO$_x$ urban	g NO$_x$	9.275	*9.133*	0.615	*0.473*	8.660	*8.660*
CO total	g CO	3.408	*3.376*	0.278	*0.246*	3.130	*3.130*
CO urban	g CO	3.294	*3.264*	0.164	*0.134*	3.130	*3.130*
PM10 total	mg PM10	344.2	*343.1*	60.2	*59.1*	284.00	*284.00*
PM10 urban	mg PM10	328.6	*327.6*	44.6	*43.6*	284.00	*284.00*
Energy Embodied	MJ LHV	13.1	*12.9*	13.1	*12.9*	0	*0*

Source: Tables 2.18 to 2.20, pp.117-118.

The same uncertainty estimates, as for LSD, were used.

Biodiesel

The feedstock for biodiesel has significant impact on the emissions. The report analysed five sources: canola, soybean, rapeseed, tallow and waste cooking oil.

Table A15. Total life cycle emissions (per MJ) calculated for biodiesel

Full Life cycle	Units	Canola biodiesel	Soybean biodiesel	Rape biodiesel	Tallow biodiesel	Tallow alternative allocation	Waste cooking oil biodiesel	Waste cooking oil alternative allocation	Uncer-tainty (%)
Greenhouse	kg CO$_2$	0.0433	0.0326	0.0443	*0.0420*	*0.0498*	*0.0062*	*0.0065*	15
NMHC total	g HC	0.145	0.172	0.146	*0.142*	*0.060*	*0.053*	*0.054*	43
NMHC urban	g HC	0.134	0.163	0.134	*0.131*	*0.059*	*0.052*	*0.053*	43
NO$_x$ total	g NO$_x$	1.296	1.283	1.314	*1.292*	*1.184*	*1.179*	*1.184*	30
NO$_x$ urban	g NO$_x$	1.219	1.235	1.221	*1.217*	*1.184*	*1.179*	*1.183*	30
CO total	g CO	0.171	0.219	0.172	*0.170*	*0.141*	*0.140*	*0.145*	72
CO urban	g CO	0.155	0.210	0.156	*0.155*	*0.141*	*0.140*	*0.144*	72
PM10 total	mg PM10	29.9	29.4	30.5	*29.8*	*27.6*	*27.5*	*27.5*	71
PM10 urban	mg PM10	28.4	28.5	28.4	*28.4*	*27.6*	*27.5*	*27.5*	71
Energy Embodied	MJ LHV	0.42	0.45	0.43	*0.41*	*0.17*	*0.14*	*0.15*	

Source: Table 4.25, p.163.

Table A16. Upstream emissions (per MJ) calculated for biodiesel

Pre-combustion	**Units**	**Canola biodiesel**	**Soybean biodiesel**	**Rape biodiesel**	**Tallow biodiesel**	**Tallow alternative allocation**	**Waste cooking oil biodiesel**	**Waste cooking oil alternative allocation**
Greenhouse	kg CO_2	0.0433	0.0326	0.0443	*0.0420*	*0.0498*	*0.0062*	*0.0065*
NMHC total	g HC	0.141	0.168	0.142	*0.138*	*0.0564*	*0.0494*	*0.0503*
NMHC urban	g HC	0.130	0.159	0.130	*0.127*	*0.055*	*0.049*	*0.049*
NO_x total	g NO_x	0.140	0.127	0.158	*0.136*	*0.028*	*0.023*	*0.027*
NO_x urban	g NO_x	0.062	0.079	0.064	*0.061*	*0.027*	*0.022*	*0.027*
CO total	g CO	0.035	0.083	0.035	*0.033*	*0.005*	*0.004*	*0.008*
CO urban	g CO	0.019	0.074	0.019	*0.019*	*0.005*	*0.004*	*0.008*
PM10 total	mg PM10	2.51	2	3.13	*2.43*	*0.219*	*0.166*	*0.166*
PM10 urban	mg PM10	1.01	1.07	1.05	*0.982*	*0.206*	*0.156*	*0.156*
Energy Embodied	MJ LHV	0.42	0.45	0.43	*0.41*	*0.17*	*0.14*	*0.15*

Source: Table 4.26, p.165.

Table A17. Tailpipe emissions (per MJ) calculated for biodiesel

Combustion	**Units**	**Biodiesel (canola)**	**Biodiesel (soybean)**	**Biodiesel (rapeseed)**	**Biodiesel (tallow)**	**Biodiesel (waste cooking oil)**
Greenhouse	kg CO_2	-	-	-	-	-
NMHC total	g HC	0.0039	0.004	0.004	*0.0038*	*0.0038*
NMHC urban	g HC	0.0039	0.0040	0.0040	*0.0038*	*0.0038*
NO_x total	g NO_x	1.156	1.156	1.156	*1.156*	*1.156*
NO_x urban	g NO_x	1.156	1.156	1.156	*1.156*	*1.156*
CO total	g CO	0.136	0.136	0.136	*0.136*	*0.136*
CO urban	g CO	0.136	0.136	0.136	*0.136*	*0.136*
PM10 total	mg PM10	27.4	27.4	27.4	*27.4*	*27.4*
PM10 urban	mg PM10	27.4	27.4	27.4	*27.4*	*27.4*
Energy Embodied	MJ LHV	0	0.000	0.000	*0.000*	*0.000*

Source: Table 4.27, p.165.

Table A18. Total life cycle emissions (per km) for trucks fuelled with biodiesel

Full Life cycle	**Units**	**Canola biodiesel**	**Soybean biodiesel**	**Rape biodiesel**	**Tallow biodiesel**	**Tallow alternative allocation**	**Waste cooking oil biodiesel**	**Waste cooking oil alternative allocation**	**Uncertainty (%) for t-km**
Greenhouse	kg CO_2	0.4310	0.3250	0.4410	*0.4180*	*0.4960*	*0.0705*	*0.0736*	15
NMHC total	g HC	1.439	1.709	1.449	*1.409*	*0.600*	*0.597*	*0.607*	71
NMHC urban	g HC	1.329	1.619	1.329	*1.299*	*0.588*	*0.587*	*0.597*	71
NO_x total	g NO_x	12.895	12.775	13.075	*12.855*	*11.784*	*11.764*	*11.814*	23
NO_x urban	g NO_x	12.125	12.292	12.144	*12.112*	*11.775*	*11.757*	*11.807*	23
CO total	g CO	1.699	2.184	1.707	*1.689*	*1.407*	*1.403*	*1.450*	106
CO urban	g CO	1.545	2.088	1.548	*1.540*	*1.404*	*1.400*	*1.447*	106
PM10 total	mg PM10	297.5	292.4	303.6	*296.7*	*274.6*	*274.3*	*274.3*	81
PM10 urban	mg PM10	282.6	283.1	282.9	*282.2*	*274.5*	*274.2*	*274.2*	81
Energy Embodied	MJ LHV	4.14	4.5	4.25	*4.05*	*1.69*	*1.61*	*1.65*	

Source: Table 4.28, p. 166; Table 4.34, p. 169.

Table A19. Upstream emissions (per km) for trucks fuelled with biodiesel

	Units	Canola biodiesel	Soybean biodiesel	Rape biodiesel	Tallow biodiesel	Tallow alternative allocation	Waste cooking oil biodiesel	Waste cooking oil alternative allocation
Greenhouse	kg CO_2	0.4310	0.3250	0.4410	*0.4180*	*0.4960*	*0.0705*	*0.0736*
NMHC total	g HC	1.4	1.67	1.41	*1.37*	*0.561*	*0.558*	*0.568*
NMHC urban	g HC	1.290	1.580	1.290	*1.260*	*0.549*	*0.548*	*0.558*
NO_x total	g NO_x	1.390	1.270	1.570	*1.350*	*0.279*	*0.259*	*0.309*
NO_x urban	g NO_x	0.620	0.787	0.639	*0.607*	*0.270*	*0.252*	*0.302*
CO total	g CO	0.343	0.828	0.351	*0.333*	*0.051*	*0.047*	*0.094*
CO urban	g CO	0.189	0.732	0.192	*0.184*	*0.048*	*0.044*	*0.092*
PM10 total	mg PM10	25	19.9	31.1	*24.2*	*2.17*	*1.87*	*1.87*
PM10 urban	mg PM10	10.1	10.6	10.4	*9.77*	*2.05*	*1.76*	*1.76*
Energy Embodied	MJ LHV	4.14	4.5	4.25	*4.05*	*1.69*	*1.61*	*1.65*

Source: Table 4.29, p. 166.

Table A20. Tailpipe emissions (per km) for trucks fuelled with biodiesel

	Units	Biodiesel (canola)	Biodiesel (soybean)	Biodiesel (rape)	Biodiesel (tallow)	Biodiesel (waste cooking oil)
Greenhouse	kg CO_2	0.000	0.000	0.000	*0.000*	*0.000*
NMHC total	g HC	0.039	0.040	0.040	*0.038*	*0.038*
NMHC urban	g HC	0.039	0.040	0.040	*0.038*	*0.038*
NO_x total	g NO_x	11.51	11.51	11.51	*11.51*	*11.51*
NO_x urban	g NO_x	11.51	11.51	11.51	*11.51*	*11.51*
CO total	g CO	1.36	1.36	1.36	*1.36*	*1.36*
CO urban	g CO	1.36	1.36	1.36	*1.36*	*1.36*
PM10 total	mg PM10	272	272	272	*272*	*272*
PM10 urban	mg PM10	272	272	272	*272*	*272*
Energy Embodied	MJ LHV	0.000	0.000	0.000	*0.000*	*0.000*

Source: Table 4.30, p. 166.

Table A21. Total life cycle emissions (per km) for buses fuelled with biodiesel

	Units	Canola biodiesel	Soybean biodiesel	Rape biodiesel	Tallow biodiesel	Tallow alternative allocation	Waste cooking oil biodiesel	Waste cooking oil alternative allocation	Uncertainty (%) for pass-km
Greenhouse	kg CO_2	0.77	0.58	0.79	*0.75*	*0.89*	*0.13*	*0.13*	7
NMHC total	g HC	2.58	3.07	2.60	*2.53*	*1.08*	*1.07*	*1.09*	15
NMHC urban	g HC	2.39	2.91	2.39	*2.33*	*1.06*	*1.05*	*1.07*	15
NO_x total	g NO_x	23.15	22.94	23.48	*23.08*	*21.16*	*21.12*	*21.21*	38
NO_x urban	g NO_x	21.77	22.07	21.81	*21.75*	*21.14*	*21.11*	*21.20*	38
CO total	g CO	3.05	3.92	3.06	*3.03*	*2.53*	*2.52*	*2.60*	37
CO urban	g CO	2.77	3.75	2.78	*2.76*	*2.52*	*2.51*	*2.60*	37
PM10 total	mg PM10	534	525	545	*533*	*493*	*493*	*493*	61
PM10 urban	mg PM10	507	508	508	*507*	*493*	*492*	*492*	61
Energy Embodied	MJ LHV	7.4	8.1	7.6	*7.3*	*3.0*	*2.9*	*3.0*	

Source: Table 4.31, p. 168; Table 4.34, p. 169.

Table A22. Upstream emissions (per km) for buses fuelled with biodiesel

	Units	Canola biodiesel	Soybean biodiesel	Rape biodiesel	Tallow biodiesel	Tallow alternative allocation	Waste cooking oil biodiesel	Waste cooking oil alternative allocation
Greenhouse	kg CO_2	0.77	0.58	0.79	*0.75*	*0.89*	*0.13*	*0.13*
NMHC total	g HC	2.51	3.00	2.53	*2.46*	*1.01*	*1.00*	*1.02*
NMHC urban	g HC	2.32	2.84	2.32	*2.26*	*0.99*	*0.98*	*1.00*
NO_x total	g NO_x	2.50	2.28	2.82	*2.42*	*0.50*	*0.47*	*0.55*
NO_x urban	g NO_x	1.11	1.41	1.15	*1.09*	*0.48*	*0.45*	*0.54*
CO total	g CO	0.62	1.49	0.63	*0.60*	*0.09*	*0.08*	*0.17*
CO urban	g CO	0.34	1.31	0.34	*0.33*	*0.09*	*0.08*	*0.16*
PM10 total	mg PM10	44.9	35.7	55.8	*43.5*	*3.9*	*3.4*	*3.4*
PM10 urban	mg PM10	18.1	19.0	18.7	*17.5*	*3.7*	*3.2*	*3.2*
Energy Embodied	MJ LHV	7.4	8.1	7.6	*7.3*	*3.0*	*2.9*	*3.0*

Source: Table 4.32, p. 168.

Table A23. Tailpipe emissions (per km) for buses fuelled with biodiesel

Combustion	Units	Biodiesel (canola)	Biodiesel (soybean)	Biodiesel (rape)	Biodiesel (tallow)	Biodiesel (waste cooking oil)
Greenhouse	kg CO_2	0.0000	0.0000	0.0000	*0.0000*	*0.0000*
NMHC total	g HC	0.070	0.071	0.071	*0.068*	*0.068*
NMHC urban	g HC	0.070	0.071	0.071	*0.068*	*0.068*
NO_x total	g NO_x	20.658	20.658	20.658	*20.658*	*20.658*
NO_x urban	g NO_x	20.658	20.658	20.658	*20.658*	*20.658*
CO total	g CO	2.434	2.434	2.434	*2.434*	*2.434*
CO urban	g CO	2.434	2.434	2.434	*2.434*	*2.434*
PM10 total	mg PM10	489.2	489.2	489.2	*489.2*	*489.2*
PM10 urban	mg PM10	489.2	489.2	489.2	*489.2*	*489.2*
Energy Embodied	MJ LHV	0.00	0.00	0.00	*0.00*	*0.00*

Source: Table 4.33, p. 169.

PULP

Table A24. Total life cycle emissions (per MJ) calculated for PULP (separate for upstream and tailpipe emissions)

	Units	WTW	WTT	TTW
Greenhouse	kg CO_2	0.0888	0.0177	0.071
HC total	g HC	0.170	0.0543	0.116
HC urban	g HC	0.141	0.026	0.116
NO_x total	g NO_x	0.185	0.094	0.091
NO_x urban	g NO_x	0.129	0.038	0.091
CO total	g CO	0.930	0.021	0.909
CO urban	g CO	0.920	0.011	0.909
PM10 total	mg PM10	38.2	5.19	33.06
PM10 urban	mg PM10	36.9	3.8	33.06
Energy embodied	MJ LHV	1.14	1.14	0

Source: Tables 12.2-12.4, pp. 327-328.

Table A25. Total life cycle emissions (per km) calculated for PULP (separate for upstream and tailpipe emissions)

	Units	PULP WTW	PULP WTT	PULP TTW
Greenhouse	kg CO_2	0.2148	0.0428	0.172
HC total	g HC	0.412	0.132	0.280
HC urban	g HC	0.342	0.062	0.280
NO_x total	g NO_x	0.447	0.227	0.220
NO_x urban	g NO_x	0.313	0.093	0.220
CO total	g CO	2.251	0.051	2.200
CO urban	g CO	2.227	0.027	2.200
PM10 total	mg PM10	92.5	12.5	80.00
PM10 urban	mg PM10	89.2	9.19	80.00
Energy embodied	MJ LHV	2.75	2.75	0

Source: Tables 12.6-12.8, p.329.

E10

Table A26. Full life cycle emissions (per MJ) calculated for E10 from various feedstock

	Units	PULP E10P (molasses-exp. sys. bound.)	PULP E10P (molasses-eco. allocat.)	PULP E10P (wheat starch waste)	PULP E10P (wheat)	PULP E10P (wheat WS)	PULP E10P (wood waste)	PULP E10P (ethylene)
Greenhouse	kg CO_2	0.0895	0.0913	0.0891	0.0911	0.0889	0.0874	0.0974
HC total	g HC	0.139	0.139	0.138	0.142	0.199	0.172	0.173
HC urban	g HC	0.111	0.112	0.111	0.112	0.168	0.145	0.141
NO_x total	g NO_x	0.175	0.174	0.173	0.185	0.181	0.170	0.186
NO_x urban	g NO_x	0.121	0.122	0.121	0.123	0.119	0.118	0.132
CO total	g CO	0.820	0.830	0.786	0.834	1.014	0.902	0.790
CO urban	g CO	0.811	0.821	0.777	0.777	0.958	0.893	0.779
PM10 total	mg PM10	38.0	38.0	39.2	39.4	40.9	39.5	38.2
PM10 urban	mg PM10	36.6	36.6	37.9	38.0	39.5	38.2	36.9
Energy embodied	MJ LHV	1.10	1.10	1.10	1.11	1.12	1.23	1.28

Source: Table 14.2, p.346.

Table A27. Upstream emissions (per MJ) calculated for E10 from various feedstocks

	Units	PULP E10P (molasses-exp.sys. bound.)	PULP E10P (molasses-eco. allocat.)	PULP E10P (wheat starch waste)	PULP E10P (wheat)	PULP E10P (wheat WS)	PULP E10P (wood waste)	PULP E10P (ethylene)
Greenhouse	kg CO_2	0.0193	0.0211	0.0189	0.0209	0.0187	0.0172	0.0227
HC total	g HC	0.0519	0.0518	0.0513	0.0554	0.112	0.0848	0.086
HC urban	g HC	0.025	0.025	0.024	0.025	0.081	0.058	0.055
NO_x total	g NO_x	0.096	0.096	0.094	0.107	0.103	0.092	0.108
NO_x urban	g NO_x	0.042	0.044	0.042	0.045	0.041	0.040	0.054
CO total	g CO	0.055	0.065	0.021	0.068	0.248	0.136	0.024
CO urban	g CO	0.045	0.055	0.011	0.012	0.192	0.127	0.014
PM10 total	mg PM10	4.93	4.9	6.19	6.38	7.89	6.49	5.19
PM10 urban	mg PM10	3.59	3.58	4.88	4.9	6.41	5.18	3.85
Energy embodied	MJ LHV	1.10	1.10	1.10	1.11	1.12	1.23	1.28

Source: Table 14.3, p.346.

Table A28. Tailpipe emissions (per MJ) calculated for E10 from various feedstocks

	Units	PULP E10P (molasses-exp. sys. bound.)	PULP E10P (molasses-eco.allocat.)	PULP E10P (wheat starch waste)	PULP E10P (wheat)	PULP E10P (wheat WS)	PULP E10P (wood waste)	PULP E10P (ethylene)
Greenhouse	kg CO_2	0.070	0.070	0.070	0.070	0.070	0.070	0.075
HC total	g HC	0.087	0.087	0.087	0.087	0.087	0.087	0.087
HC urban	g HC	0.087	0.087	0.087	0.087	0.087	0.087	0.087
NO_x total	g NO_x	0.078	0.078	0.078	0.078	0.078	0.078	0.078
NO_x urban	g NO_x	0.078	0.078	0.078	0.078	0.078	0.078	0.078
CO total	g CO	0.766	0.766	0.766	0.766	0.766	0.766	0.766
CO urban	g CO	0.766	0.766	0.766	0.766	0.766	0.766	0.766
PM10 total	mg PM10	33.06	33.06	33.06	33.06	33.06	33.06	33.06
PM10 urban	mg PM10	33.06	33.06	33.06	33.06	33.06	33.06	33.06
Energy embodied	MJ LHV	0	0	0	0	0	0	0

Source: Table 14.4, p.347.

Table A29. Total life cycle emissions (per km) calculated for E10 from various feedstocks

	Units	PULP E10P (molasses-exp. sys. bound.)	PULP E10P (molasses-eco. allocat.)	PULP E10P (wheat starch waste)	PULP E10P (wheat)	PULP E10P (wheat WS)	PULP E10P (wood waste)	PULP E10P (ethylene)
Greenhouse	kg CO_2	0.2164	0.2209	0.2157	0.2204	0.2150	0.2114	0.2358
HC total	g HC	0.336	0.335	0.334	0.344	0.481	0.415	0.418
HC urban	g HC	0.270	0.270	0.269	0.270	0.407	0.350	0.342
NO_x total	g NO_x	0.423	0.423	0.418	0.449	0.440	0.412	0.453
NO_x urban	g NO_x	0.292	0.296	0.292	0.299	0.289	0.285	0.320
CO total	g CO	1.986	2.009	1.903	2.018	2.454	2.182	1.911
CO urban	g CO	1.962	1.986	1.880	1.881	2.317	2.159	1.886
PM10 total	mg PM10	91.9	91.8	95.0	95.4	99.1	95.7	92.6
PM10 urban	mg PM10	88.7	88.7	91.8	91.9	95.5	92.5	89.3
Energy embodied	MJ LHV	2.65	2.66	2.65	2.70	2.71	2.99	3.10

Source: Table 14.6, p.347.

Table A30. Upstream emissions (per km) calculated for E10 from various feedstocks

	Units	PULP E10P (molasses-exp. sys. bound.)	PULP E10P (molasses-eco.allocat.)	PULP E10P (wheat starch waste)	PULP E10P (wheat)	PULP E10P (wheat WS)	PULP E10P (wood waste)	PULP E10P (ethylene)
Greenhouse	kg CO_2	0.0466	0.0511	0.0459	0.0506	0.0452	0.0416	0.0550
HC total	g HC	0.126	0.125	0.124	0.134	0.271	0.205	0.208
HC urban	g HC	0.060	0.060	0.059	0.060	0.197	0.140	0.132
NO_x total	g NO_x	0.233	0.233	0.228	0.259	0.250	0.222	0.263
NO_x urban	g NO_x	0.102	0.106	0.102	0.109	0.099	0.096	0.130
CO total	g CO	0.133	0.156	0.050	0.165	0.601	0.329	0.058
CO urban	g CO	0.109	0.133	0.027	0.028	0.464	0.306	0.033
PM10 total	mg PM10	11.9	11.8	15	15.4	19.1	15.7	12.6
PM10 urban	mg PM10	8.69	8.68	11.8	11.9	15.5	12.5	9.32
Energy embodied	MJ LHV	2.65	2.66	2.65	2.7	2.71	2.99	3.1

Source: Table 14.7, p.347.

Table A31. Tailpipe emissions (per km) calculated for E10 from various feedstocks

	Units	PULP E10P (molasses-exp. sys. bound.)	PULP E10P (molasses-eco.allocat.)	PULP E10P (wheat starch waste)	PULP E10P (wheat)	PULP E10P (wheat WS)	PULP E10P (wood waste)	PULP E10P (ethylene)
Greenhouse	kg CO_2	0.170	0.170	0.170	0.170	0.170	0.170	0.181
HC total	g HC	0.210	0.210	0.210	0.210	0.210	0.210	0.210
HC urban	g HC	0.210	0.210	0.210	0.210	0.210	0.210	0.210
NO_x total	g NO_x	0.190	0.190	0.190	0.190	0.190	0.190	0.190
NO_x urban	g NO_x	0.190	0.190	0.190	0.190	0.190	0.190	0.190
CO total	g CO	1.853	1.853	1.853	1.853	1.853	1.853	1.853
CO urban	g CO	1.853	1.853	1.853	1.853	1.853	1.853	1.853
PM10 total	mg PM10	80.00	80.00	80.00	80.00	80.00	80.00	80.00
PM10 urban	mg PM10	80.00	80.00	80.00	80.00	80.00	80.00	80.00
Energy embodied	MJ LHV	0	0	0	0	0	0	0

The study highlights the benefits and limitations for a higher uptake/market penetration of alternative fuels in terms of their feedstock and infrastructure availability.

Biodiesel has a high kinematic viscosity that affects the fuel atomisation during injection and hence requires modified injection systems, and also requires modifications in the refuelling systems.

Further studies are needed to investigate what the effects large-scale production of crops has on soil degradation.

Canola oil is not presently available heavy vehicle fuel due to the major alterations required in the engines to accommodate the fuel.

The major disadvantage of 100% biodiesel is related to concerns about its ability to meet Euro3 standards for PM and NO_x.

Lower sulfur fuels permit more efficient operation of emission control devices such as exhaust gas recirculation, oxidation catalysts, and particulate traps.

In assessing the health impacts of the fuels, the study reviews several schemes of weighting the pollutants: Victorian air pollution index; Stage 1 weighting system based on fuels' health effects and global warming impact; EPA (1997) weighting system based on CBA of health effects; index-based weighting, similar to air pollution index.

The authors opted for a risk-weighted scoring system based on estimates of human health risk to rank the fuels on their air pollutant emissions. On a life cycle basis, the gaseous fuels gave the lowest contribution on this criterion, followed by LSD, E95 and ULS (Table 1.3, p.375). Biodiesel scored poorly in relation to air quality because its production and use generate considerable amounts of fine particulates. The use of waste oil as a diesel extender slightly reduced greenhouse gases but increased air pollution.

The last sections of the study modelled the influence of future emission standards and investigated the impact of driving conditions.

The report made several recommendations of significance to air quality:

- more studies are necessary on the air toxics emissions and the appropriate Australian risk-weighted factors to use in examining their relative effects; and
- local data are required for aqueous and solid wastes (upstream emissions), as well as recent data for tailpipe emissions for all types of vehicles.

11. Energy Strategies (2003)

Energy Strategies conducted a LCA for CSR Sugar for the energy use and GHG emissions for the existing Sarina and the planned Burdekin distilleries. The study incorporated both the conventional allocation method (economic allocation) and the expanded system boundary method, and included updated current operational practices. Downstream emissions for petrohol (E10) and diesohol (E15) were based on data taken from an APACE study (APACE, 2003a). Table A32 shows the LCA (allocation method) emissions of GHG for E10 and the relative change compared to the conventional base fuel.

Table A32. LCA GHG emissions for E10 blended fuel

Distillery	CO_2-e (g/km) /% change wrt base fuel	CO_2-e (kg/GJ) /% change wrt base fuel
Sarina	249.2 (-3.1%)	77.1 (-2.6%)
Burdekin	238.9 (-7.1%)	73.9 (-6.6%)

Source: p 6.

The study included a comparison to the CSIRO transport fuel study (Beer *et al.,* 2001). The CSR study was based on actual operational data and stated that CSIRO analysis was not representative of the current Sarina operations due to a number of CSIRO assumptions:

- 50% of the energy required to produce azeotropic ethanol is sourced from bagasse and the balance from coal in the system boundary method (70% bagasse according to the authors);
- while 100% coal is used in the economic allocation method;
- the distillery electricity supply was sourced from bagasse combustion (not in off season as electricity is supplied from coal-fired boiler and external electricity grid.

Both studies used the same fundamental principles (ISO 14041); however, differences in application of these principles pointed out by the authors included:

- *"..present study assesses the fuel life cycle of the fuels from production through combustion";*
- allocation of emissions to bio-dunder production from distillery was not included in the CSIRO study;
- actual production and market prices of sugar and molasses was used in the upstream emission allocation to C-molasses and ethanol (allocation method) whereas CSIRO used economic allocation ratio of 97.1 sugar /2.9 molasses;
- economic values of bio-dunder and ethanol, under current market conditions, was used for the allocation ratio;
- a comparison between "with distillery" and "without distillery" was required for the expanded system boundary method analysis (not conducted by CSIRO, thereby not including the emissions and energy associated with fertiliser production from bio-dunder, alternative uses of molasses, and avoidance of production of petroleum fuel equivalent amounts, or production of "ethanol replacement").

12. General Motors Corporation, Argonne National Laboratory (2001)

This prospective study focuses on the US light-duty vehicle (LDV) market after 2005 and it compares 13 fuels, selected from 75 fuel pathways in their well-to-wheel (WTW) energy use and greenhouse emissions (GHG). Of relevance to the present study is the comparison amongst gasoline (conventional, reformulated, CARFG phase 2 and 3, Tier 2) with S contents $\in$ [5; 300 ppm], LS diesel (current diesel with 120-330 ppm S and future diesel with < 15 ppm S), neat and blended ethanol obtained from corn, woody and herbaceous biomass.

The study considers 15 vehicles (including conventional and hybrid electric vehicles) with both spark-ignition (SI) and compression-ignition (CI) engines, (as well as hybridised and non-hybridised fuel cell vehicles); the benchmark vehicle is the Chevrolet Silverado full-size pick-up.

The study determined both well-to-tank (WTT) and tank-to-wheel (TTW) emissions and provided confidence intervals for the estimates.

The LCA estimation of WTT energy use and GHG emissions was performed with GREET[2]. The findings were the following:

- the total WTT energy use for conventional (current diesel 120-350 ppm) is lower than for current gasoline, which in its turn is lower than all three types of ethanol (Figure ES-1.1, p.6);
- similarly, the upstream GHG emissions of fuel are lower for diesel (current and future) than for gasoline (Figure ES-1.4, p.10), and
- the three ethanol pathways have negative GHG emissions because of C uptake sequestration during growth of plants.

The TTW results, obtained from the vehicle simulation model) showed that the diesel CONV[3] vehicle, with compression ignition direct injection (CIDI), had a gain in fuel economy over the baseline of 18% and a TWW efficiency of 19.4, while the E85 CONV SI had no gain in fuel efficiency and 16.7 TTW efficiency (Table ES-2.1, p.16).

The TTW GHG emissions were related to the C contents of the fuel. During the integration phase, 12 out of the 30 pathways were excluded based on resource availability. One of the pathways referred to corn ethanol, which does not appear as adequate for use in high-volume transport applications.

The integrated well-to-wheel analysis showed that diesel CIDI CONV offered no energy use benefits over conventional gasoline vehicle, although the median was the lowest energy consuming value. The E85 energy pathway was higher than for gasoline.

The herbaceous E85 fuelled vehicles had the lowest GHG emissions[4]. The GHG findings show also that E85 yield to the lowest emissions/mile (< 200g), where diesel and gasoline emissions are higher than 400 g, and respectively 500 g/mile (Table ES 3.10-p. 32).

The asymmetric distributions indicate considerable opportunity for new-technology based improvements in GHG emissions for all vehicles.

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[2] For emissions, the model includes: carbon dioxide (CO_2), methane (CH_4), and nitrous oxide (N_2O), and five criteria pollutants - volatile organic compounds (VOCs), carbon monoxide (CO), nitrogen oxides (NO_x), fine particulates with diameters of 10 µm or less (PM10), and sulfur oxides (SO_x).

[3] CONV refers to a conventional vehicle, as opposed to a hybrid or an electric vehicle.

[4] The study established that the best-fit distributions for energy use and emissions were the logistic distribution for WTT and Weibull distribution for TTW estimates.

Table A33. Total life cycle energy and GHG for diesel, gasoline, and ethanol E85

WTW	Energy Use (BTU/mile)[5]			Energy share		GHG Emissions (g/mile) *(g/km)*		
	20%	**50%**	**80%**	**WTT**	**TTW**	**20%**	**50%**	**80%**
CIDI CONV DIESEL (81,245 g CO$_2$/mmBtu[6])	4,462	5,735	6,232	21%	79%	362 *225*	472 *293*	513 *319*
CONV SI Gasoline (76,477)	5,388	6,949	7,365	23	77	422	544	577
CONV SI E85 (76,289)	8,170	10,579	12,582	54	46	128	172	205

Source: pp.26-27 and Appendix 3C, p.3.29.

13. Groves (2002)

The study assessed the reductions of CO$_2$ and N$_2$O emissions, using Rapeseed Methyl Ester (RME) as a diesel-blending component in the UK. The results regarded a blend of 5% (by mass) RME in diesel, used as fuel for a Ford Focus 1.8, which was considered representative of the European vehicle fleet (vehicle energy=1.91 MJ/km and WTW energy=2.04 MJ/km, 141g CO2e/km TTW and 16g CO$_2$e/km WWT).

For calculations, Groves (2002) used an in-house model and the results were compared with the GREET model results. The reference fuel is ULSD (< 50ppm).

The study also highlighted the variability of the GHG results obtained in numerous previous studies, which was due to the model and inputs used (the overall WTW energy balance remains positive in the range 0.31 to 0.68 MJ/MJ).

The study included three scenarios (two with rapeseed cultivated in Europe and one imported from Australia) with the system boundaries covering all agricultural inputs to the production of rapeseed. The by-products of RME (straw, glycerine, oilcake) were noted together with their yields. The base-case was represented by RME production with no credits for co-products; this was compared with RME with credit for using straw in the esterification process and not NG.
The results are the following:

Table A34. Total fuel-cycle energy use and greenhouse gas emissions for rapeseed biodiesel

Scenario	g CO$_2$-e/kg fuel MJ/kg fuel	g CO$_2$/km (MJ/km) MJ/MJ fuel	g GHGe N$_2$O/MJ
RME from rapeseed grown on UK (+fertilizer)	1337[7] 37	94.2 0.51	47.1
Land conversion		207 0.5	
Import Australia		90 0.53	
Conventional diesel	3309 43	154.4 or 157? 1.15	77.2

Source: Table 9, p.20; Table 10e, p.22, p.27

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[5] 1 Btu/mile=655.6868 J/km.

[6] Fuel emission factors where mmBTU = million BTU; 1 g/Btu=0.947867 g/kJ.

[7] From 1337 g GHG, 648 are due to fertilisers and chemicals and 167 for agricultural inputs.

For the RME blend 5% (no fertiliser), the CO_2 emissions /km was 152.8 g/km (p.1).
One-way sensitivity analysis has been conducted for three parameters: production/hectare, fertilizer dose, and transport distance from field. The production had significant impact on the reduction of energy (0.61 vs. 0.41 MJ/MJ) and CO_2 (153.5 vs. 152 g/km), followed by the percentage of fertilizer.

14. Louis (2001)

The study investigated 25 combinations of fuels and vehicle technologies in their WTW energy use and GHG emissions[8]. The technologies used were internal combustion engine (ICE), hybrid, and fuel cell (FC). The fuels were assumed to be produced from either crude oil or natural gas. The calculations were performed with a slightly modified version of GREET.

The main uncertainties were represented by fuel consumption and energy efficiencies.
The study considered that 2% of the energy in oil and natural gas (NG) would be lost by flaring in the near future. This would result in a total GHG emission of 2.9 g/MJ of crude oil or NG. Venting and leaks would increase the GHG by 0.5 g/MJ CO_2-e.

The reference vehicle was a Mercedes Class A – 1.6 L engine with the energy intensity and fuel consumption of 2.42 MJ/km, respectively, 7.5 L/100km on the European Drive Cycle.
The diesel version of the A class (1.7 L) used 4.9 L/100km or 1.89MJ/km on the same drive cycle.
GHG emissions were calculated from the calorific value of the fuel and its C content.
The following table shows the results relevant for the present study:

Table A35. Total life cycle and tailpipe embodied energy and greenhouse gas emissions (g/km)

Fuel	Vehicle	Energy (MJ/km) WTW	Energy (MJ/km) TTW	GHG (g/km) WTW	GHG (g/km) TTW
Gasoline	ICE	2.84	2.42	220	172
Diesel	ICE	2.07	1.89	152	131
CNG	ICE	2.57	2.42	148	128

Source: Tables 6,7, and Appendix 2, Table 11.

The energy use and GHG emission (per km travelled) were the lowest for diesel.

15. GM-LBST (2002)

The GM-LBST study had as an objective to identify potential fuels with technical and environmental ability to complement, and eventually substitute, gasoline and diesel in the European passenger car market. It followed the American study and compared the results between the two regions.

GM-LBST investigated 44 WTT pathways (88 variants) and selected 32 pathways to derive the WTW energy use and greenhouse gas (GHG) emissions.

The fuels selected for analysis were based on crude oil (gasoline and diesel with < 10ppm S), natural gas, electricity and biomass (RME, ethanol from crop residue, poplar, and sugar beet[9]), and they were combined with the following vehicle propulsion systems: ICE, fuel cell, and hybrids, considered technically available by 2010.

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[8] N_2O is not included in calculations (reasons: small contribution and uncertainty).

[9] Distance from field to ethanol plant = 50 km, distance from ethanol plant to distribution centre/refuelling station = 150 km.

The base vehicle was the 2002 Opel Zafira minivan using 1.8 16V gasoline ICE and a 5-speed manual transmission.

The study used E^2 database model for upstream emissions and the HPSP vehicle simulation model of GM on the European Driving Cycle.

The findings were consistent with those of the American study, but absolute values were lower due to a smaller reference vehicle (minivan instead of truck/pick-up). In the upstream the higher oil refining efficiency in Europe and the EU-mix NG pathway, superior to the imported LNG American pathway, lead to greater benefits in the EU study.

In contrast with the American study, GM-LBST includes the energy content of the produced fuels. The energy losses in the European study are equivalent to the energy consumption in the American study.

The results relevant for the present study are:

- biomass-derived fuel supply pathways show the highest complexity and the widest range of results depending on applicable cultivation method, fertiliser use, soil, and climate conditions. The by-products options considered were: sugar beet as fuel, animal feed; ethanol in sugar refinery; crushed rapeseed and glycerine;
- energy use is higher for biofuels than gasoline and diesel; the lowest WTT energy use is for diesel (1.11.to 1.14 MJ/MJ), gasoline (1.14 to1.18 MJ/MJ), CNG EU Mix (1.19 to 1.26 MJ/MJ), followed by compressed methane biogas — CMG (1.5MJ/MJ) and methanol (1.54MJ/MJ), then ethanol;
- lowest WTT GHG are for CMG, ethanol, then diesel (with sulfur content < 10ppm), and CNG, gasoline and methanol; the CMG (<-50g/MJ) and ethanol emissions (poplar —30g/MJ; sugar beet -40 to 10 g/MJ) are negative as the C comes from a renewable source;
- diesel and ethanol have the better pathways than gasoline with respect to the TTW[10] greenhouse gases; and
- the GHG emissions balance is in favour of biofuels.

The energy requirements and the emissions factors used in the study are presented in the following tables (in this study we are particularly interested in DI diesel MTA and gasoline ICE MTA[11]):

Table A36. Energy requirements (TTW) for gasoline, diesel, CNG, and ethanol FC (MJ/km)

Fuel	Best estimate MJ/km	Range	Fuel	Best estimate MJ/km	Range
Gasoline (Baseline)	2.44	2.3 - 2.59	CNG	2.23	2.1 – 2.35
Diesel	1.96	1.85 – 2.11	Ethanol FPFC	1.68	1.49 – 2.19

Source: Table 2-4 p.86.

Table A37. Emission factors for fossil-fuels and biofuels (g/MJ)

Fuel	g CO2/MJ	Fuel	g CO2/MJ
Gasoline	73.4	Ethanol	71.3
Diesel	72.8	CMG	56.9
FTD	70.7	RME blended Diesel	73
CNG	56.4	RME	76.7
Blended gasoline	73.2	ETBE	71.4

Source: Table 2-5 p.87.

For CH_4 and N_2O, the amounts released per km during combustion of gasoline and diesel were:

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[10] The study did not take into consideration cold-start performance.

[11] The ethanol is studied only on FC propulsion systems, therefore the comparison for ICE involves only gasoline and diesel.

Table A38. Methane and Nitrous Oxide TTW emissions (per km) from gasoline and diesel

Vehicle	CH_4 (g/km)	N_2O (g/km)
Gasoline conventional drive SI (baseline)	0.020	0.0174
Diesel conventional drive CIDI	0.01	0.0099
Ethanol FCV	0.124	0.0035

Source: Table 2-6 pp.88-89.

By aggregating them, the total TTW exbodied GHG emissions of vehicles result as follows:

Table A39. Tailpipe greenhouse emissions (g/km)

	Best estimate (g/km)	Interval
Gasoline MTA (2010 baseline)	185	175 to 196
Gasoline 10% EBTE MTA	184	174 to 195
Direct Injection Gasoline MTA	159	147 to 181
Direct Injection Diesel MTA	146	138 to 157
DI Diesel 5% RME MTA	146	138 to 157
Ethanol FPFC	123	110 to 160

Source: Table 2-7 p.91.

Overall, the WTW embodied energy (MJ/km) and GHG emissions (g/km) of different combinations of fuel supply pathway and power train have the following values:

Table A40. Total life cycle energy requirements (MJ/km)

Fuel	Conventional MTA	Conventional Hybrid
Gasoline	2.84 (SI) – 2.45 (DI)	2.07 (SI) – 1.92 (DI)
Diesel DI	2.19	1.84

Source: Table3-2: pp.96-98.

Table A41. Total life cycle greenhouse gas emissions (g/km)

Fuel	Conventional MTA	Conventional Hybrid
Gasoline	217 (SI) – 188 (DI)	160 (SI) – 149 (DI)
Diesel DI	166	140

Source: Table3-3: pp.99-100.

Blends of 5% RME result into GHG emissions of 165 g/km (range 155 – 177, Figure 3-20, p.120), similar to diesel DI ICE MTA (range 157 – 178, Figure 3.2, p.102);
Overall, a switch from DI diesel (161 g/km) to CNG EU mix ICE MTA (165 g/km) has not proved significant GHG reductions (Figure 0-7, p.21).

Tables 1.1 (p.42) and 1.4 (p.47) provide energy input, energy losses, and GHG emissions for alternative fuels, depending on the feedstock.

16. Armstrong *et al.* (2002)

The authors updated an earlier report published in 1995 on the potential benefits of RME and ethanol as substitutes for diesel and gasoline. The report was limited to the upstream processes and did not consider their end use. The authors stressed the uncertainty in the results and presented some of the difficulties encountered in comparing different studies: type and use of by-products.

The production of biofuels involves more fuel combustion than fossil fuels and may therefore lead to an increase in combustion related pollutants such as NO_x and particulates.

In terms of the energy required to produce the biofuel, and accounting for the upstream energy of diesel and gasoline, RME and ethanol represent average savings of 47% and 17%, respectively, of the energy contained in the fuel. The figures are even larger when considering the animal feed credits (56%, 31%, respectively).

The GHG emissions are 53% less for RME and 26% less for ethanol, but with animal feed credits the savings raise to 58% and, respectively 37%.

For comparison purposes the authors used energy (Ro) and GHG balance (Rg) and conventional fuel substitution potential (So). An R figure of zero indicated a fully renewable fuel, while a value of 1 indicated no net energy saving.

17. Encinar *et al.* (2002)

Encinar *et al*. (2002) studied the trans-esterification reaction, with ethanol, of Cynara cardunculus L. oils (a Compositae plant found in Spain) using sodium and potassium hydroxides as catalysts. Compared to No. 2 diesel, the biodiesel had:

- a similar heating value (Biodiesel 40 MJ/kg and No. 2 diesel 45.2 MJ/kg);
- higher density (Biodiesel 0.87 g/L and No. 2 diesel 0.847 g/L), and
- higher cetane number (Biodiesel 48.3 to 49.1 and No. 2 diesel 46).

Cynara cardunculus appeared to be a good candidate for a renewable energy source. There are characteristics such as higher cloud and pour points that might point to potential difficulties in cold starts, but the flash and combustion points represent benefits for handling and storage (guarantee of safety).

18. Pimentel (2003)

Pimentel’s recent work published in Natural Resources Research did not share the more enthusiastic opinions of Shapouri *et al.* (2002), regarding ethanol from corn. The author stated that ethanol produced from US corn was not a renewable energy source: 20% more energy is used for producing ethanol than the energy provided by ethanol (accounting for the by-products); ethanol production increases environmental degradation due to soil erosion, use of pesticides and fertilisers; ethical priority for corn and other food crops should be for population and feed.

His findings were based on more recent data available about corn and ethanol production in the US. Pimentel (2003) showed that the Shapouri *et al.* (2002) study, used “out-of-date energy data or covering only nine states“ in the US.

When including all 50 US states, increased energy for hybrid corn and nitrogen fertilisers, and updated credit value for by-products, the outcome changed.

With respect to the environmental costs, the author underlined the need for acknowledging environmental impacts such as: soil erosion, heavy insecticide and herbicide use, nitrogen fertilisers, and problems of air and water pollution associated with the operation of ethanol plants.

The ethanol yield was about 0.37 L from 1 kg corn or the production of 1 ha (8,590 kg) can be converted into 842 gallons ethanol. The energy requirement for one ha was 33.9 million BTU, and the cost of corn feedstock only required for 1 gallon ethanol was $US 0.69. The production of ethanol (without distribution) implied a cost of $US 0.79/gallon ethanol.

Comparing the cost of producing gasoline ($US 0.63) with the cost of producing ethanol (equivalent to the energy content of gasoline) – $US 2.24, showed that the production of ethanol was uneconomical. The above costs did not include the credits for the dry distillers grains (DDG) obtained from dry-milling (- $US 0.45), and the additional costs of water pollution ($US 0.13). By including them, the ethanol cost was $US 2.15/gallon. The author drew attention to the Brazilian case — where sugarcane, a more efficient feedstock than corn, cannot change the negative energy balance. When the subsidies for the ethanol industry in Brazil were stopped, the ethanol production became economically unfeasible (p. 130).

19. Wheals *et al.* (1999)

Wheals *et al*. (1999) reflected on the 25 years of ethanol use in Brazil and North America (the two big producers of ethanol from sugar cane and corn in the world). The main motivation for the ethanol production was the reduced dependency on oil imports.

Ethanol produces lower emissions of CO and NO_x, and comparable GHG with conventional fuels.

In spite of the increased efficiency in production and crediting the by-products, the energy balance is still negative (especially when the feedstock is corn), and the commercial viability of ethanol is highly dependant on the tax credits.

The authors emphasise that the economic comparison should be based not on the distillery-gate, respectively refinery prices, but on the full fuel prices, internalising the external impacts of fuel production and combustion (Table A42).

Table A42. Costs and benefits of fuel-ethanol production

Direct costs	Direct benefits
Grain or sugarcane feedstock	Sale of ethanol
Electricity and water use	Major national agricultural industry
Transport, processing, fermentation, and distillation	Cogeneration of electricity
Distribution and supply	Sale of co-products
Tax subsidy or exemption	Technological development
Indirect costs	**Indirect benefits**
Land degradation	Reduced emissions of GHG
Increased acetaldehyde pollution	Reduced pollution by CO and NO_x
Foreign currency payments and interest	Renewable source
Military protection of gasoline supplies	Reduction in fossil-fuel use
	Reduced dependency on imported oil

Source: Wheals *et al*. (1999): 485 (Table 6).

20. Calais and Sims (2000)

Calais and Sims (2001) compared natural gas-based fuels and biofuels with gasoline and diesel in their GHG emissions. On an energy basis, the natural gas fuels were the most environmentally friendly fuels, followed by methanol and ethanol. RME biodiesel has CO_2 combustion emissions with 29% higher than diesel (Figure 1, p.4).

When accounting for the upstream emissions too, the biggest CO_2 reductions are for biodiesel (biodiesel 40%; ethanol E95 55%).

21. Roarty and Webb (2003)

The use of ethanol in Australia is small – around 135 million litres, of which 50 million are used for fuel blending. The Commonwealth Government has a policy objective of increasing output of biofuels to 350 million litres by 2010.

Fuel ethanol attracted conflicting opinions on the environmental, social, and economic benefits of its use. From tests showing possible damage to vehicles, to environmental benefits and development of regional industries, industry, press and public comments centred on the percentage of ethanol in blending.

BTRE considers that the government assistance ($0.38143/litre as excise on petrol) exceeds the environmental benefits and ABARE found that – similar to Brazil and US situations – the ethanol production is not commercially viable in Australia without assistance.

The Government announcement of a capital subsidy of $0.16/litre for new or expanded facilities for biofuels (until the capacity reaches 310 million litres or by June 2007) is viewed by the Fuel Taxation Inquiry as a misallocation of resources (p.ii).

Ethanol can be manufactured from:
- biomass via fermentation sugar derived from grain starches of many crops;
- biomass via the utilisation of ligno-cellulosic fraction of crops, and
- petroleum and natural gas via an ethylene intermediate step.

Whilst the expansion of ethanol production (at the 10% share on the fuels market) would increase the economic activity in some regional areas, industry sectors and government will lose more than $3 billion.

To enable the competition between ethanol and petrol, assistance has been provided by the government in the form of: zero rate of excise, bounty payments to producers, and from 18 September 2002, a subsidy to producers. Other forms of assistance include:
An ethanol pilot plant (funding $2 million);
- diesel and Alternative Fuels Grants Scheme ($0.20809/litre);
- $400,000 to test new fuel technology – Sugar Research Institute;
- Greenhouse Gas Abatement Programme ($7.35 million – Mosman Central Sugar Mill and $8.8 million – BP to market ethanol blend); and
- $0.16/litre capital subsidy (minimum volume 5 million and maximum $10 million/plant).

22. Van Gerpen (2000)

The author reviewed the “Comparative LCA of Biodiesel and Fossil Diesel Fuel” by Ceuterick and Spirinckx, whose findings produced some consternation in the biodiesel community.

The two authors used nine categories of environmental impacts when comparing biodiesel with diesel. Only fossil fuels (45%) and GHG (55%) were reduced for biodiesel, the remaining had higher values for biodiesel: water consumption, acidification, eutrophication, ozone, and non-radioactive and radioactive waste.

The results mainly reflect the agricultural practices in Belgium, where higher quantities of fertilisers are used for the rapeseed crops.

23. Monyem and Van Gerpen (2001)

The authors evaluated the impact of oxidised biodiesel on a John Deere 427612 turbocharged DI diesel engine in steady-state conditions at a single speed of 1400 rpm, but at two different loads (20% and 100%), and three injection timings (30 advanced, standard, and 30 retarded).
Five fuels were tested: oxidised neat soybean biofuel, unoxidised neat biofuel, diesel no.2, blend 20% oxidised, blend 20% unoxidised biofuel.

The engine performance was similar between the biodiesel, blends and diesel, but the fuel consumption was higher for biofuels (13-15%), reflecting their lower energy content.
Oxidised biodiesel produced 15% less CO than the unoxidised biofuel and 28% less than diesel no. 2 at full-load (for all injection timings). The brake-specific CO emissions for the oxidised biodiesel were 26% lower than for unoxidised and 55% lower than diesel no. 2 at 20% load.
The HC emissions were higher at the 20% load than the full-load by a factor of 10. At 20% load, the oxidised biodiesel reduced the HC emissions by 21% compared to unoxidised biodiesel (16% for full-load) and 54% compared to diesel no. 2.

The neat biodiesels produced 13-14% more NO_x than diesel no. 2. However, no statistically significant difference was found on the 20% blends and diesel no. 2. The Bosch smoke number decreased from 1.4 to 0.5 from diesel to oxidised biodiesel, but again no statistically significant different in the smoke emissions between diesel no. 2 and biodiesel blends.

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[12] Four-cylinder, four-stroke engine.

APPENDIX III. ETHANOL AS AN OCTANE ENHANCER

NAFC, OCTANE RATING AND ETHANOL

The Australian automotive industry has committed to improving the fuel efficiency of new passenger motor vehicles (NAFC) to 6.8 litres per 100 kilometres (Kemp and Mcfarlane, 2003). Meeting the targeted improvement in NAFC will most likely require the use of higher octane fuels, which may provide opportunities for increased use of ethanol in transport fuels.

The octane rating of a fuel is a measure of the fuel's tendency to knock (auto-ignite) in a test engine when compared to iso-octane. Higher-octane fuels allows for higher compression engines, increasing the fuel efficiency of spark-ignition engines.

Euro II standard petrol vehicles produced in Australia are optimised to perform using unleaded petrol (ULP), which has an octane rating of around 91 RON (Coffey, 2000, p. 2 26). Euro III compliant vehicles are optimised to perform on premium unleaded petrol (PULP) with an octane rating of 95 RON (Coffey, 2000). Some newer Euro-compliant vehicles are designed to operate on fuel with an octane rating of 98 RON, and, in anticipation of more stringent vehicle emissions and fuel standards being introduced in Europe in the future, it is reasonable to expect an increasing proportion of European vehicles will be optimised to operate on 98 RON fuel. The requirement that new petrol-fuelled vehicles sold in Australia meet Euro III vehicle emissions standards from 2005 is expected to increase the demand for 95 RON petrol in the future. It is anticipated that if, and when, Euro IV (petrol) vehicle emissions standards are introduced in Australia, 95 RON petrol will be the 'standard' automotive fuel.

Enhancing the octane rating of automotive fuels

There are two principal ways by which the octane rating of petrol may be increased:

- additional refinery processing to convert low octane components into higher octane components, using a combination of isomerisation, alkylation and reforming[13]; or
- through the use of chemical additives, typically an alcohol (such as ethanol), ether or organometallic compound.

Currently refinery processes, a mix of isomerisation, alkylation and reformulation, are employed for converting low octane components into higher octane components. EA (2000a) note, however, that the ability of isomerisation and alkylation to lift octane is limited, and it is likely that Australian refiners will have to rely on reforming, i.e. increasing the level of aromatics, to increase the octane rating. There are also limits on the aromatic content of automotive fuels in place through the *Fuel Standards (Petrol) Determination 2001*, possibly limiting the extent to which reforming may be used and providing an opportunity for the increased use of ethanol in Australian petrol.

Ethanol as an Octane Enhancer

Ethanol is one of a number of compounds that may be used in petrol as an oxygenate and octane enhancer. Other compounds that may be used to enhance octane include MTBE, ETBE and methanol. As already noted, up until now Australian refineries have generally used refinery processes to convert low octane components into higher ones, thereby enhancing the overall octane rating of the fuel.

Ethanol is probably one of the few alcoholic or etheric oxygenates currently permitted for blending with petrol. Under the current fuel standards, MTBE, DIPE and TBA are effectively prohibited from use in petrol in Australia. A range of other oxygenates, ETBE, TAME and ETAE, are, according to

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[13] For the domestic refining industry importation of higher octane fuel supplies is also an option.

DSA (2000), not listed on the Australian Inventory of Chemical Substances, consequently they are not permitted to be used in Australia.

In many countries, MTBE is the preferred oxygenate of the oil industry. For example, MTBE is used widely in Europe and Asia (DEH, 2000) to enhance octane rating. MTBE is produced by the addition of methanol to olefin isobutene, a by-product of refinery and petrochemical operations. Typically MTBE can be blended with petrol in concentrations of up to 15%.

Concerns about the environmental effects of MTBE contamination of groundwater have led to a ban on the use of MTBE in transport fuels in the US, and to Australian standards that will effectively ban fuel containing MTBE in Australia from 1 January 2004. DSA (2000) states that the problem concerning the use of MTBE in petrol is fundamentally related to the failure to properly transport and store petrol. DSA (2000) also notes that it is unlikely that Europe could adopt the new Euro III and Euro IV emissions standards without the use of MTBE in petrol.

Among the alcohol compounds that may be used to enhance octane, ethanol and iso-propyl alcohol (IPA) are among the best for raising existing 91 RON rated petrol to 95 RON rating (DSA, 2000). DSA (2000, table 4, p. 19) provides some estimates of the impact on RON of adding varying concentrations of octane enhancing compounds to 1998 pool petrol, with an initial octane rating of 91.6/82.5 (RON/MON). Ethanol and IPA provide the biggest lift in RON per unit volume of additive, lifting the RON to 93.9 and 94.1, respectively. Some additional refining, to lift the RON of the base petrol stock would be required to ensure that the ethanol blend fuel obtained 95 RON. The addition of 10% ethanol to petrol lifts the oxygen content of the fuel to 3.5% by weight, which is at the limit allowed for ethanol blended fuels under the current fuel standards.

Issues Associated With the Use of Ethanol as an Octane Enhancer

Any decision by refiners to use ethanol to enhance the octane rating of automotive fuels, would have to take account of issues such as cost and supply, reliability, environmental performance and marketability (consumer acceptance).

Cost and Supply

According to a report by Coffey Geosciences (2000), it is technically feasible for Australia to adopt Euro III petrol standards (i.e. 95 RON and 150 ppm sulfur) without the use of octane enhancing compounds. For refiners, the choice between refinery based octane enhancement and additives will thus depend on relative production costs. Increased production of 95 RON petrol at the refinery would likely require additional capital investment and slightly higher operating costs. Coffey Geosciences (2000) estimated that the cost of moving to fuel standards for Euro III compliant vehicles would result in an average increase in fuel prices of 0.5 cents per litre, with ongoing production costs of 0.15 cents per litre.

Volume of supply is also an issue. Most alcohols are constrained in supply, which limits their likely use as an octane enhancer. For example, the use of 10% ethanol in all petrol used in Australia would require domestic production and/or import of around 1,900 ML of anhydrous ethanol. At present, total domestic production capacity is around 80–105 ML. It is unlikely that octane enhancing compounds would be required for all PULP sold, as some Australian refineries apparently have the capacity to increase production of PULP without the use of added compounds and with little additional investment. However, total fuel ethanol production would probably have to expand significantly to be considered for widespread use as an octane enhancer.

Demand for 95 RON petrol is likely to increase significantly by 2009–10 as a result of the vehicle emissions standards and government initiatives. In particular, it is expected that the introduction of Euro III vehicle standards from 2005–06, together with the Government's announced excise incentives to speed the transition to PULP will increase the demand for higher octane fuels. Based on these initiatives, the BTRE (2003) projects that the demand for 95 RON petrol will increase from around 1 billion litres in 2002–03 to around 2.2 billion litres in 2009–10.

Uptake of 95 RON fuel could be accelerated if Euro IV (petrol) vehicle emissions standards are implemented. In adopting the Euro IV vehicle emissions standards, 95 RON fuel is likely to be required to enable vehicles to meet mandated emission standards from 2008 onwards. Moreover, it is unlikely that vehicle manufacturers could meet the NAFC targets without using 95 RON fuel. Meeting Euro IV production of standard fuels, with the lower sulfur content, are likely to entail additional processing costs. Coffey Geosciences (2003) has estimated that the additional costs to domestic refiners of producing Euro IV standard fuel (95 RON and 50 ppm sulfur) would entail capital costs of $A175 million per refinery (although the cost would vary across refineries) and additional operating costs of $A17 million per annum per refinery. Retail prices would increase by around 1.1–1.2 cents per litre.

Most Euro III compliant vehicles are generally optimised for 95 RON petrol, with some designed for 98 RON petrol (Coffey, 2003, p. 50). Adoption of Euro IV vehicle standards would increase the proportion of vehicles that require 98 RON petrol. DSA (2000) suggest that any move to mass produce 98 RON fuels in Australia, however, would probably require the use of octane enhancing additives in the fuel.

Environmental Concerns

Groundwater contamination is not restricted to MTBE. All alcoholic and etheric oxygenates, including ethanol, can contaminate ground water. MTBE, however, is of particular concern as it has a much longer half-life (>120 days) than many other oxygenates (e.g. ethanol 4.1 days).

The addition of ethanol to petrol can raise the volatility, as measured by the Reid Vapour Pressure (RVP), of the blended fuel. A 10% ethanol blend fuel will raise the RVP of the base petrol stock from 60.0 kPa to 74.6 kPa. Increases in the vapour pressure can significantly increase the rate of evaporative emissions. For example, FORS (1997) has estimated that a reduction in the vapour pressure from 77 kPa to 70 kPa would reduce evaporative emissions by as much as 55%. In most States there are restrictions on the volatility of fuels. In some States, such as New South Wales, State agencies have agreements with fuel refiners to reduce the volatility of transport fuels in summer months. Volatility of an ethanol blend fuel may be addressed by reducing the volatility of the base petrol stock.

Consumer Confidence

A critical issue for refiners is consumer confidence in ethanol blend fuels. Prior to 1 July 2003, fuel retailers have not had to reveal the ethanol content of fuel sold to motorists. The *Fuel Standard (Petrol) Amendment Determination 2003 (No 1)* caps the volume of ethanol that may be blended with petrol to 10%. In addition, the Minster for the Environment and Heritage has made the Fuel Quality Information Standard (Ethanol) Determination 2003 that will require the labelling of petrol containing ethanol from 1 March 2004. Until the introduction of these legislative instruments, the ethanol content of ethanol blend fuels was not limited and consumers knew little about the energy content of the fuel when refuelling their vehicles.

Recently, public confidence in ethanol has been adversely affected by reports of engine damage from the use of ethanol blends exceeding 10%. Prior to legislation to cap the ethanol content of fuel, the ethanol content of ethanol blends in some areas exceeded 10%. Tests of blends of up to 10% ethanol exhibit no adverse impact on vehicle performance.

The mandatory labelling of ethanol blended fuel and 10% cap on ethanol in petrol, will give consumers more reliable information about the price/quality mix of fuel purchases and may allay consumer concerns about ethanol blend fuels. The reference case biofuels consumption projections, presented in the report, assume that consumer confidence in ethanol blends of 10% or less will recover and all of the domestically produced ethanol available for the transport market will be used in E10.

Implications for Transport Analysis

Because all of these issues would need to be resolved before refiners considered ethanol as a part of any strategy for meeting increased demand for higher octane automotive fuels, it appears too early to say definitively that ethanol would be widely used as a general octane enhancer. Consequently, for the transport analysis the study assumes that ethanol is used purely as a fuel extender and that there is no increase in ethanol use as a result of the introduction of new vehicle emissions or fuel quality standards.

APPENDIX IV. FUEL QUALITY AND VEHICLE EMISSIONS STANDARDS

This appendix provides details of the proposed fuel quality and vehicle emissions standards that will be introduced over the next 5–10 years.

FUEL QUALITY STANDARDS

The *Fuel Quality Standards Act 2000* (FQSA) introduced a legislative framework for the assurance of national fuel quality standards for petrol and diesel for the first time in Australia. The new standards are prescribed in *the Fuel Standard (Petrol) Determination 2001,* and the *Fuel Standard (Automotive Diesel) Determination 2001*, which have been made under the *Fuel Quality Standards Act 2000* (FQSA). The FQSA and fuel standard determinations provide the framework for the harmonisation of Australian fuel quality standards with international standards. Notable features of the currently mandated standards are that they prohibit the supply of leaded petrol and reduce the level of sulfur in diesel and petrol. Standards are presently in place for petrol, diesel, LPG and biodiesel. A standard for CNG is being examined.

Standards are also to be put in place for biofuels. As mentioned, there is already a standard in place covering biodiesel, the *Fuel Standard (Biodiesel) Determination 2003*, and the Department of the Environment and Heritage has begun the process of preparing a discussion paper on a proposed fuel quality standard for fuel ethanol (both for use in blending with petrol and as pure fuel ethanol) in Australia.

Petrol Standards

The proposed fuel standards for each of the different grades of petrol, and the date of application, are listed in Table A43.

The Minister for the Environment and Heritage gazetted an amendment to the determination, *Fuel Standard (Petrol) Amendment Determination 2003 (No. 1)*. This Determination, which commenced on 1 July 2003, caps the volume of ethanol that can be blended with petrol at 10%. Recent amendments to fuel quality standards, enacted through the *Fuel Quality Standards Amendment Act 2003*, provide the Minister with the power to require the mandatory labelling of specified fuels. Under the legislation, from 1 March 2004, retailers will be required to specify whether the fuel contains ethanol.

Table A43. Fuel standards – Petrol

Property	Grade	Amount	Date
Sulfur	ULP, LRP	500 mg/kg	1 Jan 2002
Sulfur	PULP	150 mg/kg	1 Jan 2002
Sulfur	All grades	150 mg/kg	1 Jan 2005
Benzene	All grades	1% volume by volume	1 Jan 2006
Lead	All grades	0.005 g/L	1 Jan 2002
Oxygen	(a) All grades of petrol not containing ethanol	2.7% mass by mass (maximum)	1 Jan 2002
	(b) All grades of petrol containing ethanol	3.5% mass by mass (maximum)	1 Jul 2003
Ethanol	All grades	10% volume by volume (maximum)	1 Jul 2003
Phosphorus	ULP, PULP	0.0013 g/L	1 Jan 2002
DIPE	All grades	1% volume by volume	1 Jan 2002
MTBE	All grades	1% volume by volume	1 Jan 2004
TBA	All grades	0.5% volume by volume	1 Jan 2002
Distillation	FBP 210°C (max)	All grades	1 Jan 2005
Olefins	18% pool average over 6 months with a cap of 20%	All grades	1 Jan 2004
	18% max by vol		1 Jan 2005
Aromatics	45% pool average over 6 months with a cap of 48%	All grades	1 Jan 2002
	42% pool average over 6 months with a cap of 45%		1 Jan 2005
MON	85.0 (min)	PULP	16 Oct 2002
	81.0 (min)	ULP	16 Oct 2002
	82.0 (min)	LRP	16 Oct 2002
Copper corrosion (3 hrs @ 50^0C)	Class 1 (max)	All	16 Oct 2002
Existent Gum (washed)	50 mg/L (max)	All	16 Oct 2002
Induction Period	360 minutes (min)	All	16 Oct 2002

Notes DIPE – Di-isoproprpyl ether (CAS no. 108-20-3).
MTBE – Methyl tertiary-butyl ether (CAS no. 1634-04-4).
TBA – Tertiary butyl alcohol (CAS no. 75-65-0).
Source: *Fuel Standard (Petrol) Determination 2001.*

Automotive Diesel Standards

The *Fuel Standard (Automotive Diesel) Determination 2001* requires that diesel that contains a substance mentioned in Table A44 must not contain more than the amount mentioned for the substance from the date mentioned.

Table A44. Fuel standards – Automotive diesel

Property	Amount	Date
Sulfur	500 mg/kg	31 Dec 2002
Sulfur	50 mg/kg	1 Jan 2006
Ash and suspended solids	100 mg/kg	1 Jan 2002
PAH	11% mass by mass	1 Jan 2006
Cetane Index	46 (min) index	1 Jan 2002
Density	820 to 860 kg/m^3	1 Jan 2002
	820 to 850 kg/m^3	1 Jan 2006
Distillation T95	370°C (max)	1 Jan 2002
	360°C (max)	1 Jan 2006
Viscosity	2.0 to 4.5 cSt @ 40°C	1 Jan 2002
Carbon Residue (10% distillation residue)	0.2 mass% max	16 Oct 2002
Water and sediment	0.05 vol% max	16 Oct 2002
Conductivity @ambient temp	50 pS/m (Min) @ambient temp (only applies at terminals, refineries, major distribution centres)	16 Oct 2002
Oxidation Stability	25 mg/L max	16 Oct 2002
Colour	2 max	16 Oct 2002
Copper Corrosion (3 hrs @50^{0}C)	Class 1 max	16 Oct 2002
Flash point	61.5^{0}C min	16 Oct 2002
Filter blocking tendency	2.0 max	16 Oct 2002
Lubricity	0.460 mm (max) (only for diesel containing less than 500 ppm sulfur)	16 Oct 2002

Notes PAH – Polycyclic aromatic hydrocarbons.
Sources: *Fuel Standard (Automotive Diesel) Determination 2001.*

Biodiesel Fuel Standards

The *Fuel Standard (Biodiesel) Determination 2003* requires that biodiesel that contains a substance mentioned in Table A45 must not contain more than the amount mentioned for the substance from the date mentioned.

Table A45. Fuel standards – Biodiesel

Property	Amount	Date
Sulfur	50 mg/kg	18 September 2003
Sulfur	10 mg/kg	1 February 2006
Sulfated ash	0.020% mass	18 September 2003
Carbon residue – 10% distillation residue; or	0.30% mass	18 September 2003
Carbon residue – 100% distillation sample	0.050% mass	18 September 2003
Water and sediment	0.050% vol	18 September 2003
Phosphorus	10 mg/kg	18 September 2003
Free glycerol	0.020% mass	18 September 2004
Total glycerol	0.250% mass	18 September 2004
Metals – Group I (Na, K)	5 mg/kg	18 September 2004
Metals – Group II (Ca, Mg)	5 mg/kg	18 September 2004
Alcohol	0.20% (m/m)	18 September 2004

Notes PAH – Polycyclic aromatic hydrocarbons.
Sources: *Fuel Standard (Biodiesel) Determination 2003.*

VEHICLE EMISSIONS STANDARDS

Future Australian vehicle emissions standards are controlled through the Australian Design Rules—ADR 37 *Emission Control for Light Vehicles* and ADR 70 *Exhaust Emission Control for Diesel Engined Vehicles*. The Australian standards involve the gradual adoption of European vehicle emissions standards for petrol and diesel fuelled vehicles. For petrol fuelled vehicles the standards and implementation dates are:

- Euro II in 2003–04 for all new petrol vehicles; and
- Euro III in 2005–06 for all new petrol vehicles;

and for diesel fuelled vehicles the standards and implementation dates are:

- Euro II in 2002–03 for all new diesel vehicles;
- Euro III in 2002–03 for all new medium and heavy duty diesel vehicles, and
- Euro IV in 2006–07 for all new diesel vehicles.

European vehicle emissions standards also encompass the introduction of tighter emissions standards for petrol fuelled vehicles from 2005 (Euro IV) and lower NO_x emissions from heavy-duty diesel vehicles from 2008 (proposed Euro V diesel vehicle emissions standards). The introduction of Euro IV (petrol) and Euro V (diesel) vehicle standards in Australia are currently under consideration by the MVEC (MVEC, 2003). There is no timetable yet for the introduction of Euro V diesel vehicle emissions standards in Australia.

Table A46 lists the emission standards for petrol fuelled vehicles and Table A47 lists the emissions limits for diesel fuelled vehicles.

Table A46. Comparison of Passenger Car (Petrol) Emission Standards

Standard	**European introduction**	**Emissions limits**			
		CO (g/km)	**HC (exhaust) (g/km)**	**NO_x (g/km)**	**HC (evaporative) (g/test)**
ADR37/01[a] UN ECE	1997–99	2.1	0.26	0.63	2
Euro II[b]	1996	2.2	0.2		2
Euro III[c]	2000	2.3	0.2	0.15	2
Euro IV[d]	2005	1.0	0.1	0.08	2

[a] The Australian standard (ADR37/01) requires the emission limits to be met for a period of 5yrs/80,000km and the test method is the same as that used in the US standard.

[b] The *Euro II* and *Euro III* standards require the emission limits to be met for a period of 5yrs/80,000km.

[c] CO limit for *Euro III* is nominally higher, but *Euro II* test excludes the first forty seconds of testing from sampling, thus making the CO limit much harder to meet

[d] The *Euro IV* standards require the emission limits to be met for a period of 5yrs/100,000km.

Sources: DOTARS (1999, table 1, p. 9).

Table A47. Comparison of 'Heavy Duty' (Diesel) Vehicle Emission Standards

Standard	**European introduction**	**Emissions limits (g/kWh)**			
		CO	**HC**	**NO_x**	**PM**
ADR37/01[a]	1997–99	4.5	1.1	8.0	0.36
Euro II[b]	1996	4.0	1.1	7.0	0.15
Euro III[c]	2000				
ESC limit		2.1	0.66	5.0	0.10
ETC limit		5.45	0.78	5.0	0.16
Euro IV	2005				
ESC limit		1.5	0.46	3.5	0.02
ETC limit		4.0	0.55	3.5	0.03
Euro V	2008				
ESC limit		1.5	0.46	2.0	0.02
ETC limit		4.0	0.55	2.0	0.03

Note ESC – European Stationary Cycle test, ETC – European Transient Cycle test.

[a] Original Euro II limit for PM was 0.25, which was reduced to 0.15 in 1998.

[b] Non-methane hydrocarbons.

[c] Smaller engines are subject to more relaxed PM limits of 0.13 (ESC) and 0.21 (ETC).

Sources DOTARS (1999, table 2, p. 10) and DieselNet (http://www.dieselnet.com/).

VEHICLE EMISSIONS STANDARDS FUEL QUALITY REQUIREMENTS

Table A48 illustrates the proposed timetable for the introduction of the tighter vehicle emissions and fuel quality standards, together with proposed implementation dates for changes to domestic fuel excise and assistance arrangements. The move to Euro III (petrol) vehicle standards will require fuel with sulfur content limited to 150ppm, aromatics be not more than 45% and an octane rating of 95 RON . Introduction of Euro IV (petrol) vehicle standards in Australia (which are due to be adopted in Europe in 2005 (MVEC 2003)), would require reductions in sulfur content to 50ppm and aromatic content to 35%, but no change to the octane content would be required; 95 RON fuel is adequate for Euro IV petrol vehicle standards (MVEC, 2003). The European Parliament has mandated that the sulfur content of both petrol and diesel fuels should be no more than 10ppm from 1 January 2009 (with member states required to make quantities available from 2005). Australia will require that from 1 January 2006, all diesel fuel sold have a sulfur content of no more than 50ppm.

Table A48. Chronology of proposed vehicle emissions and fuel quality standards

Year – 1 July	Petrol vehicles	Diesel vehicles	Alternative fuels and biofuels
2001			
2002		Euro II all new diesel vehicles; Euro III for all new medium and heavy duty diesel vehicles	
2003	10% cap on ethanol Euro II emissions standards for all new petrol vehicles	Additional 1c/L excise – regular diesel	
2004		Additional 1c/L excise – regular diesel	
2005	Euro III emissions standards for all new petrol vehicles	sulfur 150ppm	
2006	Additional 0.06c/L excise on all petrol LRP/ULP (until 30-Jun-2008). Production subsidy of 1.1c/L for 50ppm sulfur PULP (until 30-Jun-2008).	Euro IV emissions standards for all new diesel vehicles sulfur 50ppm	
2007		Additional 0.7 c/L excise on all diesel (until 30-Jun-2009). Production subsidy of 1.0 c/L for diesel 10ppm sulfur content (until 30-Jun-2009).	
2008			Commencement of phased introduction of excise on all untaxed fuels. Commencement of offsetting production subsidies for untaxed fuels.
2009			
2010			
2011			
2012			Full excise on all fuels. Termination of offsetting production subsidies on alternative fuels.

APPENDIX V. EXCISE DECISIONS OF 16 DECEMBER 2003

PRIME MINISTER

FUEL EXCISE REFORMS

Today I announce an overhaul of the fuel excise system. These reforms will result in a more consistent and neutral tax regime for fuels used in vehicles.

The government announced in the 2003-04 Budget that excise rates will be set for all fuels capable of being used in an internal combustion engine. It has been decided that no excise will apply to new fuels until 1 July 2008. Excise will then be introduced in five equal annual steps to a final rate on 1 July 2012.

For administrative simplicity, a banded excise system will be adopted, with differing rates for high, medium and low energy fuels. Excise on diesel and petrol remains unchanged under the banded system.

Alternative fuels entering the excise net will receive a discount of 50% on the full energy content rate. Excise on petrol and diesel will remain at 38.143 cents per litre (c/L). Excise on LPG and ethanol will be 12.5 c/L – half their energy content value.

Excise will not be levied on domestic (eg heating and cooking) uses of gaseous fuels such as LPG. Business use of gaseous fuels (LPG, CNG, LNG) in non-transport applications (eg power generation) will also be effectively excise free.

The 50% discount applying to alternative fuels entering the excise net was set having regard to a range of industry, regional and other factors.

The government has also announced additional assistance for the LPG sector to assist its transition into the excise net. A $1000 subsidy will be made available from 1 July 2008 – the point at which excise first applies to LPG at 2.5 c/L - to consumers who purchase a dedicated or dual fuel LPG vehicle that is delivered new. This subsidy will be available for three years.

The excise reforms announced today represent a first tranche in the development of a comprehensive package of transport-related measures, including broad-based reform of the business excise credit system and future road funding. Details of further reforms are being developed and will be announced in 2004.

Final rates for fuels (including discounts) are shown in Table 1 below. The transition path for fuels entering the excise net is shown at Table 2 below.

Table A49. New excise rates

Fuel type	Energy Content **(MJ/L)**	EXCISE RATE **(c/L)**	ALTERNATIVE FUELS **(c/L)**
High-energy content fuels Petrol, diesel, biodiesel, GTL diesel	**Above 30**	38.143	19.1 **(biodiesel)**
Mid-energy content fuels Eg. LPG, LNG, ethanol, dimethyl ether	**Between 20 – 30**	25	12.5 **(LPG, ethanol, LNG)**
Low-energy content fuels Eg. Methanol	**Below 20**	17	8.5 **(methanol)**
Other: Eg. CNG	**Between 38 – 41** **{MJ/m^3}**	38 **{c/m^3}**	19 **{c/m^3 }**

Table A50. Transition path for fuels entering the excise net

Fuel type	1 July 2003	1 July 2004	1 July 2005	1 July 2006	1 July 2007	1 July 2008	1 July 2009	1 July 2010	1 July 2011	1 July 2012
High-energy content **Biodiesel**	**0**	**0**	**0**	**0**	**0**	3.8	7.6	11.4	15.3	**19.1**
Mid-energy content **LPG, LNG, ethanol**	**0**	**0**	**0**	**0**	**0**	2.5	5.0	7.5	10.0	**12.5**
Low-energy content **Methanol**	**0**	**0**	**0**	**0**	**0**	1.7	3.4	5.1	6.8	**8.5**
Other **CNG (cents per m^3)**	**0**	**0**	**0**	**0**	**0**	3.8	7.6	11.4	15.2	**19.0**

16 December 2003

APPENDIX VI. FUEL LIFE CYCLE ENVIRONMENTAL IMPACTS OF BIOFUELS

ENVIRONMENTAL PERFORMANCE OF ETHANOL COMPARED TO UNLEADED PETROL (ULP)

Fuel Life Cycle GHG Emissions per km

Table A51. Fuel life cycle greenhouse gases (in g per km) from E10 and ULP (passenger car)

Impact category	Unit	E10 (ULP) (molasses cogen energy)	E10 (ULP) (molasses)	E10 (ULP) (sorghum)	E10 (ULP) (wheat)	E10 (ULP) (wheat starch waste)	ULP
CO_2 (Upstream)	g CO_2	58.84	64.65	69.62	70.58	65.75	56.21
Methane (Upstream)	g CH_4	0.4933	0.5001	0.5213	0.5203	0.5087	0.5431
N_2O (Upstream)	g N_2O	0.004704	0.0048	-0.003462	0.01044	0.000481	0.000329
Other	g CO_2-e	0	7.85E-06	8.50E-06	8.49E-06	8.49E-06	0
CO_2 (Tailpipe)	g CO_2	316.5	316.5	316.5	316.5	316.5	340.5
Methane (Tailpipe)	g CH_4	0.007421	0.007421	0.007421	0.007421	0.007421	0.007159
N_2O (Tailpipe)	g N_2O	0.002269	0.002269	0.002269	0.002269	0.002269	0.002271
CO_2	g CO_2	375.34	381.15	386.12	387.08	382.25	396.71
Methane	g CH_4	0.500721	0.507521	0.528721	0.527721	0.516121	0.550259
N_2O	g N_2O	0.006973	0.007069	-0.001193	0.012709	0.00275	0.0026
GHG (Upstream)	g CO_2-e	70.65754	76.640108	79.4940885	84.74271	76.58166	67.71712
GHG (Tailpipe)	g CO_2-e	317.35923	317.35923	317.359231	317.3592	317.3592	341.3543
GHG total	g CO_2-e	388.01677	393.99934	396.8533195	402.1019	393.9409	409.0715

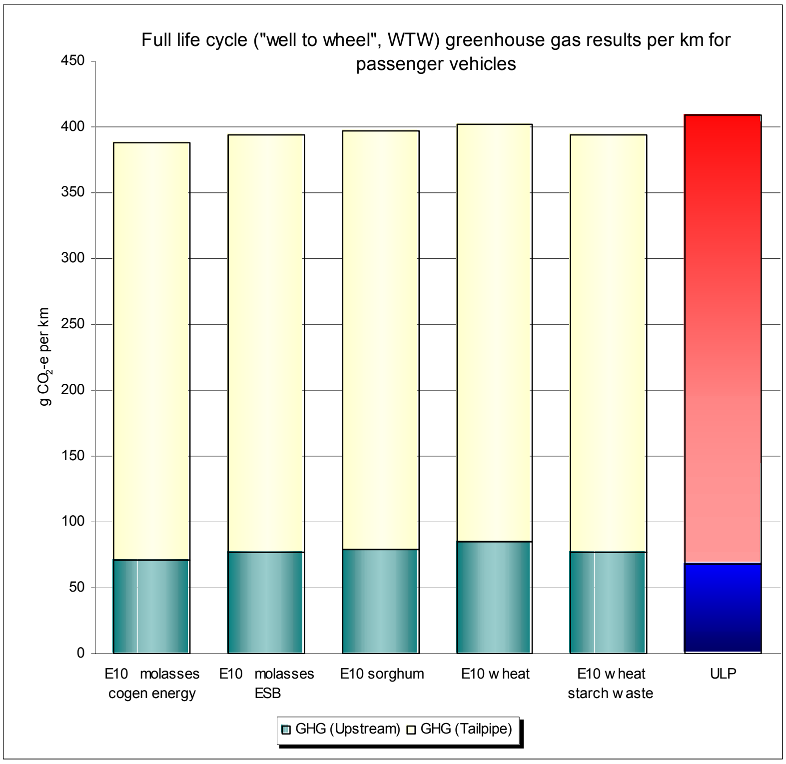

Figure A13. Fuel life cycle greenhouse gases from E10 and ULP separate for upstream and tailpipe (passenger car)[14]

Fuel Life Cycle Air Pollutant Emissions per km

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[14] Upstream emissions are always shown in the bottom part of the histogram bars. The ULP results are shown in a different colour to emphasise that it was the reference fuel.

Table A52. Full life cycle air pollutants (in g per km) from E10 and ULP (passenger car)

Impact category	Unit	E10 (ULP) (molasses cogen energy)	E10 (ULP) (molasses)	E10 (ULP) (sorghum)	E10 (ULP) (wheat)	E10 (ULP) (wheat starch waste)	ULP
CO (Tailpipe)	g CO	3.547	3.547	3.547	3.547	3.547	4.85
CO (Upstream)	g CO	0.2876	0.2873	0.1005	0.3625	0.1021	0.09036
NO_x (Tailpipe)	g NO_x	0.4846	0.4846	0.4846	0.4846	0.4846	0.4614
NO_x (Upstream)	g NO_x	0.4733	0.5021	0.4867	0.543	0.4834	0.4802
NMVOC (Tailpipe)	g NMVOC	0.1437	0.1437	0.1437	0.1437	0.1437	0.1678
NMVOC (Upstream)	g NMVOC	0.6833	0.6814	0.6803	0.6997	0.6799	0.6692
Particulates (Tailpipe)	mg PM10	3.343	3.343	3.343	3.343	3.343	3.346
Particulates (Upstream-Urban)	mg PM10	6.692	13.19	13.96	13.93	13.75	7.062
Particulates (Upstream-Non-urban)	mg PM10	7.025	7.007	6.307	7.557	6.757	7.442
CO total	g CO	3.8346	3.8343	3.6475	3.9095	3.6491	4.94036
NO_x total	g NO_x	0.9579	0.9867	0.9713	1.0276	0.968	0.9416
NMVOC total	g NMVOC	0.827	0.8251	0.824	0.8434	0.8236	0.837
Particulates All	mg PM10	17.06	23.54	23.61	24.83	23.85	17.85

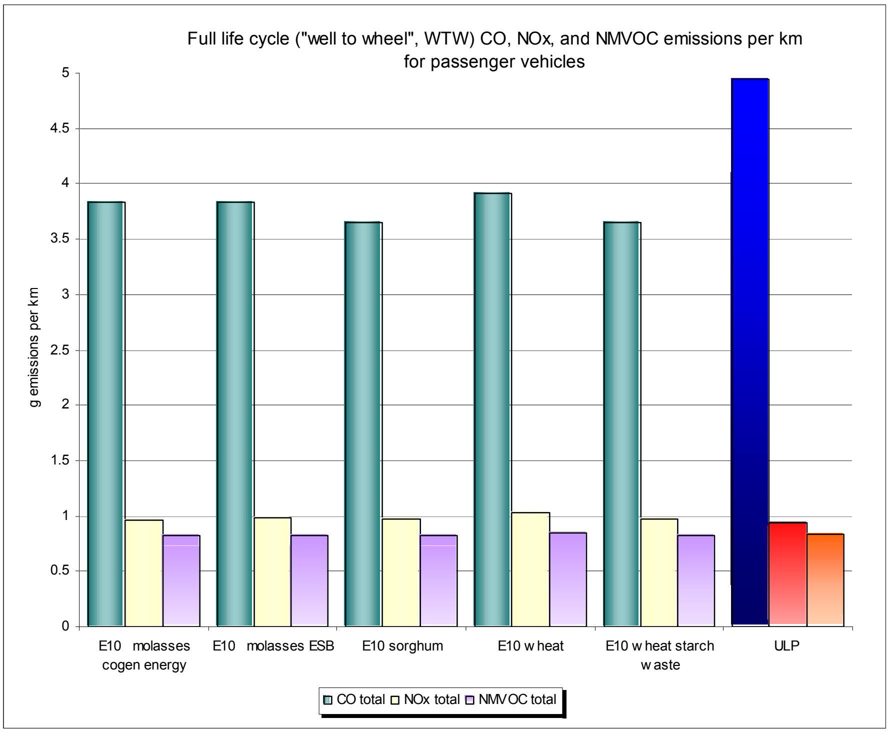


Figure A14. Full life cycle CO, NO$_x$, and NMVOC emissions from E10 and ULP (passenger car)

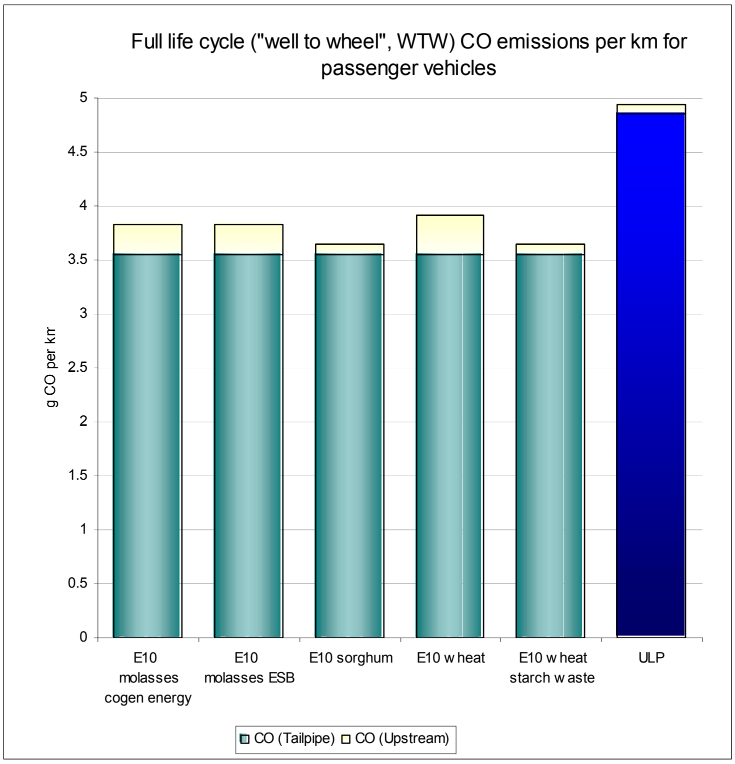


Figure A15. Full life cycle CO emissions from E10 and ULP separate for upstream tailpipe emissions (passenger car)

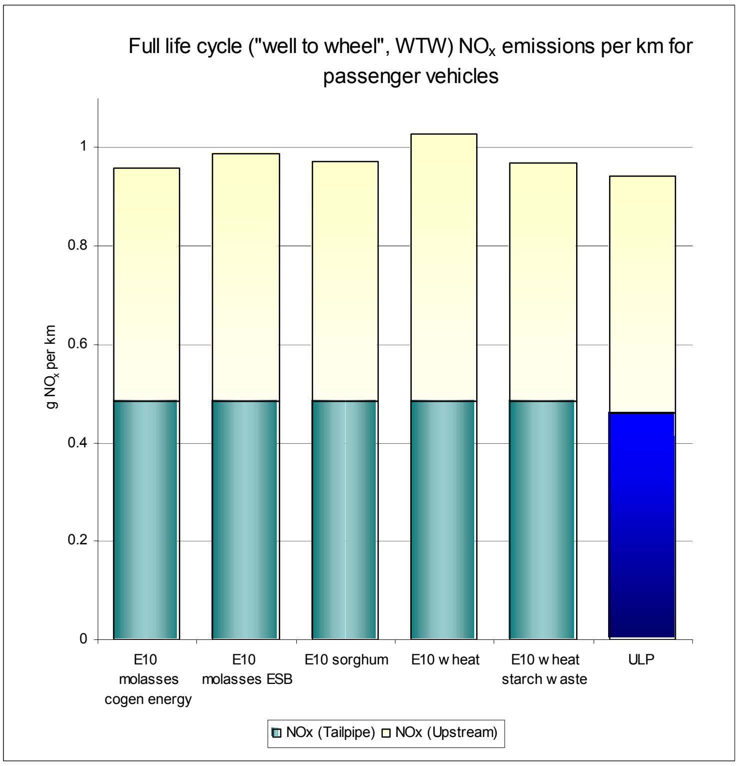
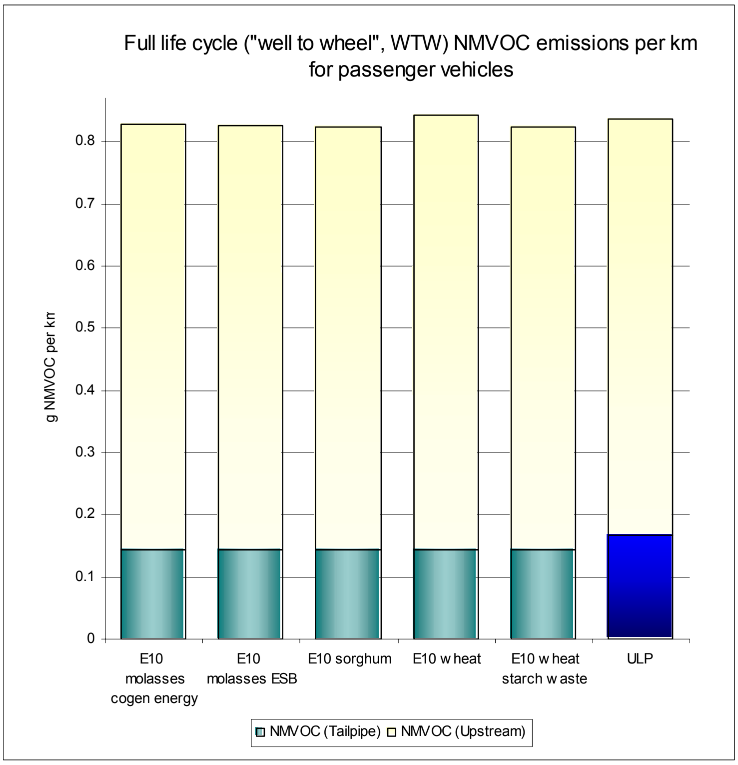


Figure A16. Full life cycle NO_x emissions from E10 and ULP separate for upstream tailpipe emissions (passenger car)

Figure A17. Full life cycle NMVOC emissions from E10 and ULP separate for upstream tailpipe emissions (passenger car)



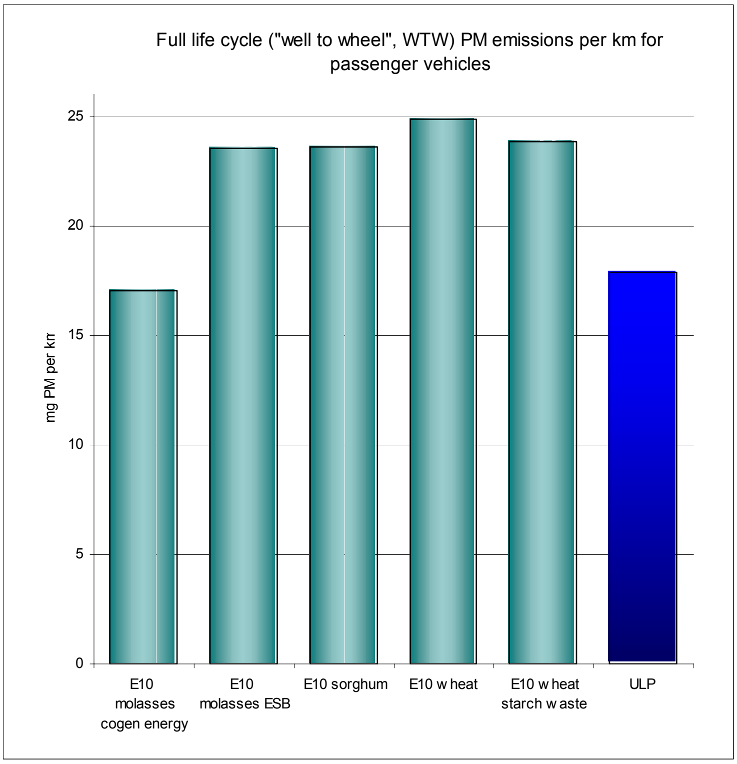


Figure A18. Full life cycle PM emissions from E10 and ULP (passenger car)

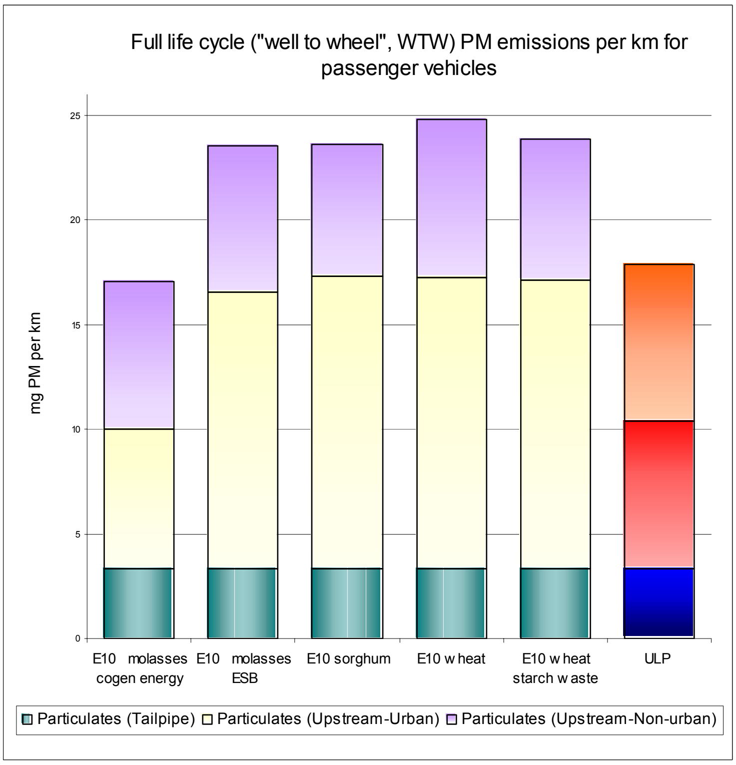


Figure A19. Full life cycle PM emissions from E10 and ULP separate for upstream-urban, upstream-non urban and tailpipe emissions (passenger car)

ENVIRONMENTAL PERFORMANCE OF BIODIESEL COMPARED TO LSD, ULS AND XLS

Full Life Cycle Greenhouse Gas Emissions per km (Rigid Trucks)

Table A53. Fuel life cycle greenhouse gas emissions from BD100, LSD, ULS, and XLS separate for upstream tailpipe emissions (rigid truck)

Impact category	Unit	Biodiesel (canola) BD100	Biodiesel (tallow) BD100	Biodiesel (waste oil) BD100	LS diesel	ULS diesel	XLS diesel
CO_2 (Upstream)	g CO_2	451.3	410	92.27	145.6	164.2	181.4
Methane (Upstream)	g CH_4	1.084	1.002	0.162	1.249	1.243	1.234
N_2O (Upstream)	g N_2O	0.923	0.8676	0.00048	0.000851	0.001044	0.00123
CO_2 (Tailpipe)	g CO_2	0	0	0	816.4	798.6	785.8
Methane (Tailpipe)	g CH_4	0.198	0.198	0.198	0.224	0.2178	0.2144
N_2O (Tailpipe)	g N_2O	0.0163	0.0163	0.0163	0.01787	0.01754	0.01727
CO_2	g CO_2	451.3	410	92.27	962	962.8	967.3
Methane	g CH_4	1.282	1.2	0.36	1.473	1.4608	1.4484
N_2O	g N_2O	0.9393	0.8839	0.01678	0.018721	0.01859	0.0185
GHG (Upstream)	g CO_2-e	760.194	699.998	95.82068	172.0927	190.6266	207.6941
GHG (Tailpipe)	g CO_2-e	9.211	9.211	9.211	826.6437	808.6112	795.6561
GHG total	g CO_2-e	769.405	709.209	105.0317	998.7364	999.2378	1,003.35

Table A54. Fuel life cycle greenhouse gas emissions from BD20 canola and corresponding diesel base fuels, separate for upstream tailpipe emissions (rigid truck)

Impact category	Unit	Biodiesel BD20 (canola & LSD)	Biodiesel BD20 (canola & ULS)	Biodiesel BD20 (canola & XLS)	LS diesel	ULS diesel	XLS diesel
CO_2 (Upstream)	g CO_2	195.6	213.5	231.1	145.6	164.2	181.4
Methane (Upstream)	g CH_4	1.188	1.206	1.223	1.249	1.243	1.234
N_2O (Upstream)	g N_2O	0.1632	0.1633	0.1633	0.000851	0.001044	0.00123
CO_2 (Tailpipe)	g CO_2	631.7	624.2	619.4	816.4	798.6	785.8
Methane (Tailpipe)	g CH_4	0.2053	0.2041	0.1921	0.224	0.2178	0.2144
N_2O (Tailpipe)	g N_2O	0.01681	0.01681	0.01681	0.01787	0.01754	0.01727
CO_2	g CO_2	827.3	837.7	862.8	962	962.8	967.3
Methane	g CH_4	1.3933	1.4101	1.4151	1.473	1.4608	1.4484
N_2O	g N_2O	0.18001	0.18011	0.18011	0.01872	0.01859	0.0185
GHG (Upstream)	g CO_2-e	271.14	289.449	307.406	172.0927	190.6266	207.6941
GHG (Tailpipe)	g CO_2-e	641.222	633.697	628.645	826.6437	808.6112	795.6561
GHG total	g CO_2-e	912.3624	923.146	936.051	998.7364	999.2378	1,003.35

Table A55. Table 10.5 Fuel life cycle greenhouse gas emissions from BD20 tallow and corresponding diesel base fuels, separate for upstream tailpipe emissions (rigid truck)

Impact category	Unit	Biodiesel BD20 (tallow & LSD)	Biodiesel BD20 (tallow & ULS)	Biodiesel BD20 (tallow & XLS)	LS diesel	ULS diesel	XLS diesel
CO_2 (Upstream)	g CO_2	188.3	206.2	223.9	145.6	164.2	181.4
Methane (Upstream)	g CH4	1.173	1.192	1.209	1.249	1.243	1.234
N_2O (Upstream)	g N_2O	0.1534	0.1536	0.1535	0.000851	0.001044	0.00123
CO_2 (Tailpipe)	g CO_2	631.7	624.2	619.4	816.4	798.6	785.8
Methane (Tailpipe)	g CH4	0.2053	0.2041	0.1921	0.224	0.2178	0.2144
N_2O (Tailpipe)	g N_2O	0.01681	0.01681	0.01681	0.01787	0.01754	0.01727
CO_2	g CO_2	820	830.4	843.3	962	962.8	967.3
Methane	g CH_4	1.3783	1.3961	1.4011	1.473	1.4608	1.4484
N_2O	g N_2O	0.17021	0.17041	0.17031	0.018721	0.01859	0.0185
GHG (Upstream)	g CO_2-e	260.487	278.848	296.874	172.0927	190.6266	207.6941
GHG (Tailpipe)	g CO_2-e	641.2224	633.697	628.645	826.6437	808.6112	795.6561
GHG total	g CO_2-e	901.7094	912.545	925.519	998.7364	999.2378	1,003.35

Table A56. Table 10.6 Fuel life cycle greenhouse gas emissions from BD20 waste oil and corresponding diesel base fuels, separate for upstream tailpipe emissions (rigid truck)

Impact category	Unit	Biodiesel BD20 (waste oil & LSD)	Biodiesel BD20 (waste oil & ULS)	Biodiesel BD20 (waste oil & XLS)	LS diesel	ULS diesel	XLS diesel
CO_2 (Upstream)	g CO_2	132.4	150.3	168	145.6	164.2	181.4
Methane (Upstream)	g CH4	1.025	1.044	1.061	1.249	1.243	1.234
N_2O (Upstream)	g N_2O	0.000763	0.000937	0.00111	0.000851	0.001044	0.00123
CO_2 (Tailpipe)	g CO_2	631.7	624.2	619.4	816.4	798.6	785.8
Methane (Tailpipe)	g CH4	0.2053	0.2041	0.1921	0.224	0.2178	0.2144
N_2O (Tailpipe)	g N_2O	0.01681	0.01681	0.01681	0.01787	0.01754	0.01727
CO_2	g CO_2	764.1	774.5	787.4	962	962.8	967.3
Methane	g CH_4	1.2303	1.2481	1.2531	1.473	1.4608	1.4484
N_2O	g N_2O	0.017573	0.017747	0.01792	0.018721	0.01859	0.0185
GHG (Upstream)	g CO_2-e	154.162	172.515	190.625	172.0927	190.6266	207.6941
GHG (Tailpipe)	g CO_2-e	641.222	633.697	628.645	826.6437	808.6112	795.6561
GHG total	g CO_2-e	795.384	806.212	819.27	998.7364	999.2378	1,003.35

Table A57. Fuel life cycle greenhouse gas emissions from BD5 canola and corresponding diesel base fuels, separate for upstream tailpipe emissions (rigid truck)

Impact category	Unit	Biodiesel BD5 (canola & LSD)	Biodiesel BD5 (canola & ULS)	Biodiesel BD5 (canola & XLS)	LS diesel	ULS diesel	XLS diesel
CO_2 (Upstream)	g CO_2	157.4	176.9	199.6	145.6	164.2	181.4
Methane (Upstream)	g CH_4	1.228	1.247	1.271	1.249	1.243	1.234
N_2O (Upstream)	g N_2O	0.04132	0.0391	0.04168	0.000851	0.001044	0.00123
CO_2 (Tailpipe)	g CO_2	776.4	758.9	752.8	816.4	798.6	785.8
Methane (Tailpipe)	g CH_4	0.2191	0.2142	0.214	0.224	0.2178	0.2144
N_2O (Tailpipe)	g N_2O	0.01775	0.0175	0.0175	0.01787	0.01754	0.01727
CO_2	g CO_2	933.8	935.8	952.4	962	962.8	967.3
Methane	g CH_4	1.4471	1.4612	1.4852	1.473	1.4608	1.4484
N_2O	g N_2O	0.05907	0.0566	0.05918	0.018721	0.01859	0.0185
GHG (Upstream)	g CO_2-e	195.997	219.208	239.212	172.0927	190.6266	207.6941
GHG (Tailpipe)	g CO_2-e	786.504	768.823	762.723	826.6437	808.6112	795.6561
GHG total	g CO_2-e	982.5008	984.031	1,001.94	998.7364	999.2378	1,003.35

Table A58. Fuel life cycle greenhouse gas emissions from BD5 tallow and corresponding diesel base fuels, separate for upstream tailpipe emissions (rigid truck)

Impact category	Unit	Biodiesel BD5 (tallow & LSD)	Biodiesel BD5 (tallow & ULS)	Biodiesel BD5 (tallow & XLS)	LS diesel	ULS Diesel	XLS diesel
CO_2 (Upstream)	g CO_2	155.5	176.9	197.8	145.6	164.2	181.4
Methane (Upstream)	g CH_4	1.224	1.247	1.267	1.249	1.243	1.234
N_2O (Upstream)	g N_2O	0.03889	0.0391	0.03925	0.000851	0.001044	0.00123
CO_2 (Tailpipe)	g CO_2	776.4	758.9	752.8	816.4	798.6	785.8
Methane (Tailpipe)	g CH_4	0.2191	0.2142	0.214	0.224	0.2178	0.2144
N_2O (Tailpipe)	g N_2O	0.01775	0.0175	0.0175	0.01787	0.01754	0.01727
CO_2	g CO_2	931.9	935.8	950.6	962	962.8	967.3
Methane	g CH_4	1.4431	1.4612	1.4812	1.473	1.4608	1.4484
N_2O	g N_2O	0.05664	0.0566	0.057	0.018721	0.01859	0.0185
GHG (Upstream)	g CO_2-e	193.2599	215.208	236.575	172.0927	190.6266	207.6941
GHG (Tailpipe)	g CO_2-e	786.5036	768.823	762.723	826.6437	808.6112	795.6561
GHG total	g CO_2-e	979.7635	984.03	999.298	998.7364	999.2378	1,003.35

Table A59. Fuel life cycle greenhouse gas emissions from BD5 waste oil and corresponding diesel base fuels, separate for upstream tailpipe emissions (rigid truck)

Impact category	Unit	Biodiesel BD5 (waste oil & LSD)	Biodiesel BD5 (waste oil & ULS)	Biodiesel BD5 (waste oil & XLS)	LS diesel	ULS diesel	XLS diesel
CO_2 (Upstream)	g CO_2	141.6	162.9	183.9	145.6	164.2	181.4
Methane (Upstream)	g CH_4	1.187	1.21	1.23	1.249	1.243	1.234
N_2O (Upstream)	g N_2O	0.000825	0.001031	0.001236	0.000851	0.001044	0.00123
CO_2 (Tailpipe)	g CO_2	776.4	758.9	752.8	816.4	798.6	785.8
Methane (Tailpipe)	g CH_4	0.2191	0.2142	0.214	0.224	0.2178	0.2144
N_2O (Tailpipe)	g N_2O	0.01775	0.0175	0.0175	0.01787	0.01754	0.01727
CO_2	g CO_2	918	921.8	936.7	962	962.8	967.3
Methane	g CH_4	1.4061	1.4242	1.4442	1.473	1.4608	1.4484
N_2O	g N_2O	0.018575	0.01853	0.01874	0.01872	0.01859	0.0185
GHG (Upstream)	g CO_2-e	166.7827	188.63	210.113	172.0927	190.6266	207.6941
GHG (Tailpipe)	g CO_2-e	786.5036	768.823	762.723	826.6437	808.6112	795.6561
GHG total	g CO_2-e	953.2863	957.453	972.836	998.7364	999.2378	1,003.35

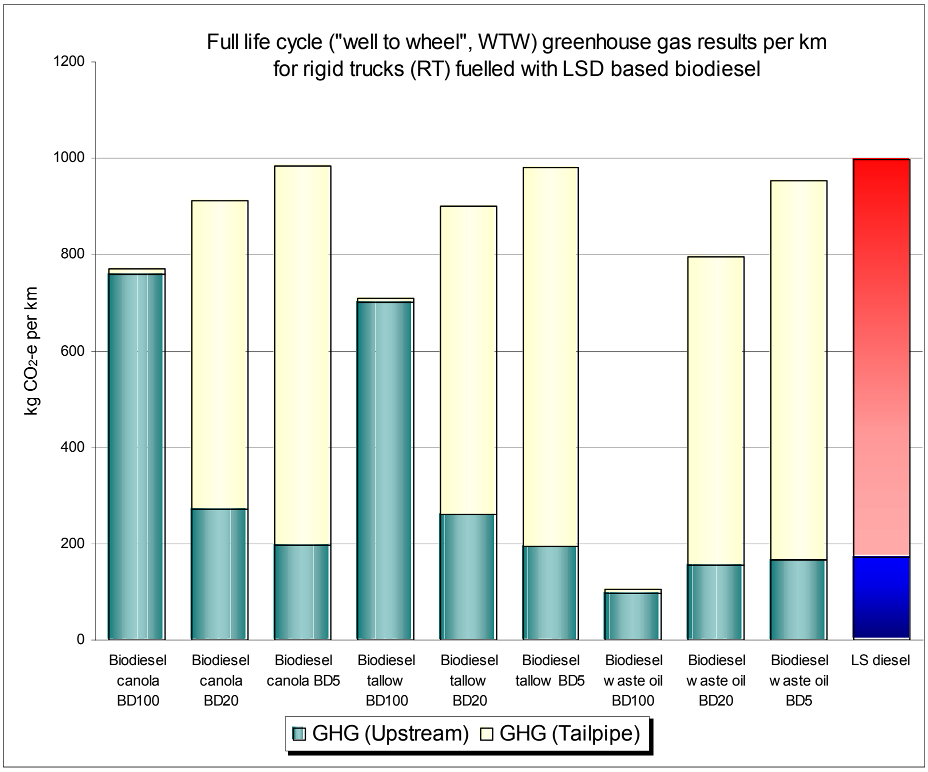


Figure A20. Fuel life cycle GHG emissions from biodiesel BD100, BD20, and BD5, compared to LSD (rigid truck)

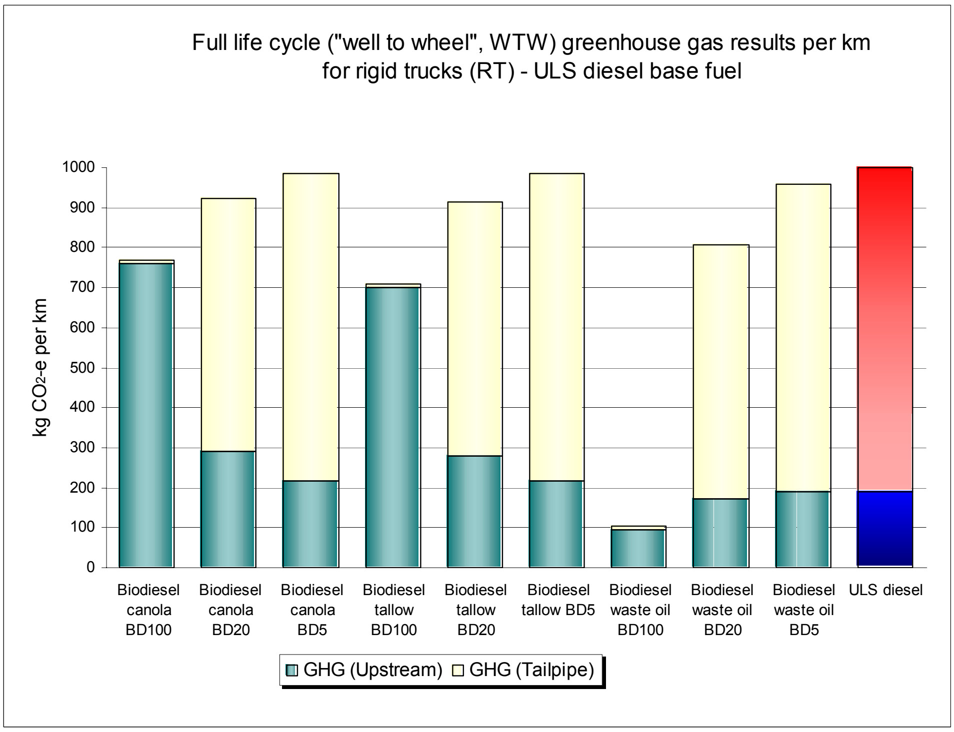


Figure A21. Fuel life cycle GHG emissions from biodiesel BD100, BD20, and BD5, compared to ULS (rigid truck)

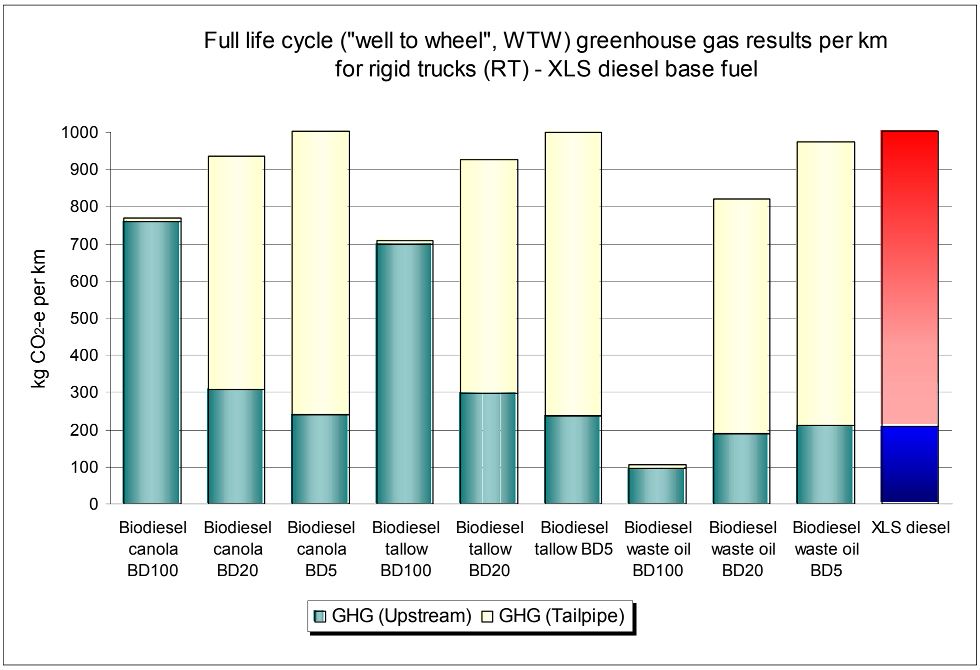


Figure A22. **Fuel life cycle GHG emissions from biodiesel BD100, BD20, and BD5, compared to XLS (rigid truck)**

Full Life Cycle Air Pollutant Emissions per km (Rigid Trucks)

Table A60. **Fuel life cycle air pollutant emissions from BD100, LSD, ULS, and XLS separate for upstream tailpipe emissions (rigid truck)**

Impact category	Unit	Biodiesel (canola) BD100	Biodiesel (tallow) BD100	Biodiesel (waste oil) BD100	LS diesel	ULS diesel	XLS diesel
CO (Tailpipe)	g CO	1.743	1.743	1.743	3.305	3.267	3.192
CO (Upstream)	g CO	0.8901	0.5481	0.1785	0.3234	0.352	0.3782
NO_x (Tailpipe)	g NO_x	12.34	12.34	12.34	12.24	10.89	9.528
NO_x (Upstream)	g NO_x	1.9	1.722	0.3485	1.165	1.303	1.431
NMVOC (Tailpipe)	g NMVOC	0.705	0.705	0.705	1.04	0.9075	0.8337
NMVOC (Upstream)	g NMVOC	0.3206	0.2779	0.05508	0.4731	0.4806	0.4865
Particulates (Tailpipe)	mg PM10	283.1	283.1	283.1	432.1	338.8	321.6
Particulates (Upstream-Urban)	mg PM10	17.48	16.01	2.457	17.25	17.97	18.6
Particulates (Upstream-Non-urban)	mg PM10	16.22	15.09	0.443	16.75	16.53	16.1
CO total	g CO	2.6331	2.2911	1.9215	3.6284	3.619	3.5702
NO_x total	g NO_x	14.24	14.062	12.6885	13.405	12.193	10.959
NMVOC total	g NMVOC	1.0256	0.9829	0.76008	1.5131	1.3881	1.3202
Particulates total	mg PM10	316.8	314.2	286	466.1	373.3	356.3

Table A61. Fuel life cycle air pollutant emissions from 20% canola biodiesel compared to LSD, ULS, and XLS separate for upstream tailpipe emissions (rigid truck)

Impact category	Unit	Biodiesel BD20 (canola & LSD)	Biodiesel BD20 (canola & ULS)	Biodiesel BD20 (canola & XLS)	LS diesel	ULS diesel	XLS diesel
CO (Tailpipe)	g CO	2.596	2.593	2.593	3.305	3.267	3.192
CO (Upstream)	g CO	0.4148	0.4441	0.4729	0.3234	0.352	0.3782
NO_x (Tailpipe)	g NO_x	11.51	11.1	10.8	12.24	10.89	9.528
NO_x (Upstream)	g NO_x	1.264	1.399	1.532	1.165	1.303	1.431
NMVOC (Tailpipe)	g NMVOC	0.7803	0.7563	0.7203	1.04	0.9075	0.8337
NMVOC (Upstream)	g NMVOC	0.4339	0.4489	0.4635	0.4731	0.4806	0.4865
Particulates (Tailpipe)	mg PM10	366.2	322.9	300.8	432.1	338.8	321.6
Particulates (Upstream-Urban)	mg PM10	16.84	17.75	18.64	17.25	17.97	18.6
Particulates (Upstream-Non-urban)	mg PM10	16.26	16.35	16.36	16.75	16.53	16.1
CO total	g CO	3.0108	3.037	3.0659	3.6284	3.619	3.5702
NO_x total	g NO_x	12.774	12.499	12.332	13.405	12.193	10.959
NMVOC total	g NMVOC	1.2142	1.2052	1.1837	1.5131	1.3881	1.3202
Particulates total	mg PM10	399.3	357	335.8	466.1	373.3	356.3

Table A62. Fuel life cycle air pollutant emissions from 20% tallow biodiesel compared to LSD, ULS, and XLS

Impact category	Unit	Biodiesel BD20 (tallow & LSD)	Biodiesel BD20 (tallow & ULS)	Biodiesel BD20 (tallow & XLS)	LS diesel	ULS diesel	XLS diesel
CO (Tailpipe)	g CO	2.596	2.593	2.593	3.305	3.267	3.192
CO (Upstream)	g CO	0.3545	0.3839	0.4128	0.3234	0.352	0.3782
NO_x (Tailpipe)	g NO_x	11.51	11.1	11.08	12.24	10.89	9.528
NO_x (Upstream)	g NO_x	1.233	1.367	1.5	1.165	1.303	1.431
NMVOC (Tailpipe)	g NMVOC	0.7803	0.7563	0.7203	1.04	0.9075	0.8337
NMVOC (Upstream)	g NMVOC	0.4264	0.4414	0.4559	0.4731	0.4806	0.4865
Particulates (Tailpipe)	mg PM10	366.2	322.9	300.8	432.1	338.8	321.6
Particulates (Upstream-Urban)	mg PM10	16.58	17.49	18.38	17.25	17.97	18.6
Particulates (Upstream-Non-urban)	mg PM10	16.12	16.11	16.22	16.75	16.75	16.53
CO total	g CO	2.9505	2.9769	3.005	3.6284	3.619	3.5702
NO_x total	g NO_x	12.743	12.467	12.3	13.405	12.193	10.959
NMVOC total	g NMVOC	1.2067	1.1977	1.1761	1.5131	1.3881	1.3202
Particulates total	mg PM10	398.9	356.5	335.4	466.1	373.3	356.3

Table A63. Fuel life cycle air pollutant emissions from 20% waste oil biodiesel compared to LSD, ULS, and XLS

Impact category	Unit	Biodiesel BD20 (waste oil & LSD)	Biodiesel BD20 (waste oil & ULS)	Biodiesel BD20 (waste oil & XLS)	LS diesel	ULS diesel	XLS diesel
CO (Tailpipe)	g CO	2.596	2.593	2.593	3.305	3.267	3.192
CO (Upstream)	g CO	0.2895	0.3188	0.3478	0.3234	0.352	0.3782
NO_x (Tailpipe)	g NO_x	11.51	11.1	11.08	12.24	10.89	9.528
NO_x (Upstream)	g NO_x	0.991	1.126	1.259	1.165	1.303	1.431
NMVOC (Tailpipe)	g NMVOC	0.7803	0.7563	0.7203	1.04	0.9075	0.8337
NMVOC (Upstream)	g NMVOC	0.3872	0.4022	0.4168	0.4731	0.4806	0.4865
Particulates (Tailpipe)	mg PM10	366.2	322.9	300.8	432.1	338.8	321.6
Particulates (Upstream-Urban)	mg PM10	14.2	15.11	16	17.25	17.97	18.6
Particulates (Upstream-Non-urban)	mg PM10	13.5	13.59	13.6	16.75	16.75	16.53
CO total	g CO	2.8855	2.9118	2.9408	3.6284	3.619	3.5702
NO_x total	g NO_x	12.501	12.266	12.059	13.405	12.193	10.959
NMVOC total	g NMVOC	1.1675	1.1585	1.137	1.5131	1.3881	1.3202
Particulates total	mg PM10	393.9	351.6	330.4	466.1	373.3	356.3

Table A64. Fuel life cycle air pollutant emissions from 5% canola biodiesel compared to LSD, ULS, and XLS

Impact category	Unit	Bio diesel BD5 (canola & LSD)	Bio diesel BD5 (canola & ULS)	Bio diesel BD5 (canola & XLS)	LS diesel	ULS diesel	XLS diesel
CO (Tailpipe)	g CO	2.763	2.754	2.754	3.305	3.267	3.192
CO (Upstream)	g CO	0.3446	0.3797	0.4139	0.3234	0.352	0.3782
NO_x (Tailpipe)	g NO_x	11.69	11.63	10.65	12.24	10.89	9.528
NO_x (Upstream)	g NO_x	1.184	1.345	1.502	1.165	1.303	1.431
NMVOC (Tailpipe)	g NMVOC	0.8201	0.7956	0.7589	1.04	0.9075	0.8337
NMVOC (Upstream)	g NMVOC	0.4611	0.4791	0.4963	0.4731	0.4806	0.4865
Particulates (Tailpipe)	mg PM10	419.7	331.7	320.7	432.1	338.8	321.6
Particulates (Upstream-Urban)	mg PM10	17.07	18.09	19.2	17.25	17.97	18.6
Particulates (Upstream-Non-urban)	mg PM10	16.63	16.65	16.7	16.75	16.75	16.53
CO	g CO	3.1076	3.1337	3.1679	3.6284	3.619	3.5702
NO_x	g NO_x	12.874	12.975	12.152	13.405	12.193	10.959
NMVOC	g NMVOC	1.2812	1.747	1.2552	1.5131	1.3881	1.3202
Particulates total	mg PM10	453.4	366.5	356.6	466.1	373.3	356.3

Table A65. Fuel life cycle air pollutant emissions from 5% tallow biodiesel compared to LSD, ULS, and XLS

Impact category	Unit	Bio diesel BD5 (tallow & LSD)	Bio diesel BD5 (tallow & ULS)	Bio diesel BD5 (tallow & XLS)	LS diesel	ULS diesel	XLS diesel
CO (Tailpipe)	g CO	2.763	2.754	2.754	3.305	3.267	3.192
CO (Upstream)	g CO	0.3296	0.3646	0.3989	0.3234	0.352	0.3782
NO_x (Tailpipe)	g NO_x	11.69	11.63	10.65	12.24	10.89	9.528
NO_x (Upstream)	g NO_x	1.176	1.337	1.494	1.165	1.303	1.431
NMVOC (Tailpipe)	g NMVOC	0.8201	0.7956	0.7589	1.04	0.9075	0.8337
NMVOC (Upstream)	g NMVOC	0.4592	0.4773	0.4944	0.4731	0.4806	0.4865
Particulates (Tailpipe)	mg PM10	419.7	331.7	320.7	432.1	338.8	321.6
Particulates (Upstream-Urban)	mg PM10	17	18.09	19.14	17.25	17.97	18.6
Particulates (Upstream-Non-urban)	mg PM10	16.5	16.61	16.66	16.75	16.75	16.53
CO	g CO	3.0926	3.1186	3.1529	3.6284	3.619	3.5702
NO_x	g NO_x	12.866	12.967	12.144	13.405	12.193	10.959
NMVOC	g NMVOC	1.2793	1.2729	1.2533	1.5131	1.3881	1.3202
Particulates total	mg PM10	453.2	366.4	356.5	466.1	373.3	356.3

Table A66. Fuel life cycle air pollutant emissions from 5% waste oil biodiesel compared to LSD, ULS, and XLS

Impact category	Unit	Bio diesel BD5 (waste oil & LSD)	Bio diesel BD5 (waste oil & ULS)	Bio diesel BD5 (waste oil & XLS)	LS diesel	ULS diesel	XLS diesel
CO (Tailpipe)	g CO	2.763	2.754	2.754	3.305	3.267	3.192
CO (Upstream)	g CO	0.3134	0.3484	0.3827	0.3234	0.352	0.3782
NO_x (Tailpipe)	g NO_x	11.69	11.63	10.65	12.24	10.89	9.528
NO_x (Upstream)	g NO_x	1.116	1.276	1.434	1.165	1.303	1.431
NMVOC (Tailpipe)	g NMVOC	0.8201	0.7956	0.7589	1.04	0.9075	0.8337
NMVOC (Upstream)	g NMVOC	0.4494	0.4675	0.4847	0.4731	0.4806	0.4865
Particulates (Tailpipe)	mg PM10	419.7	331.7	3,209.7	432.1	338.8	321.6
Particulates (Upstream-Urban)	mg PM10	16.41	17.49	18.55	17.25	17.97	18.6
Particulates (Upstream-Non-urban)	mg PM10	15.89	16.01	16.05	16.75	16.75	16.53
CO	g CO	3.0764	3.1024	3.1367	3.6284	3.619	3.5702
NO_x	g NO_x	12.806	12.906	12.084	13.405	12.193	10.959
NMVOC	g NMVOC	1.2695	1.2631	1.2436	1.5131	1.3881	1.3202
Particulates total	mg PM10	452	365.2	355.3	466.1	373.3	356.3

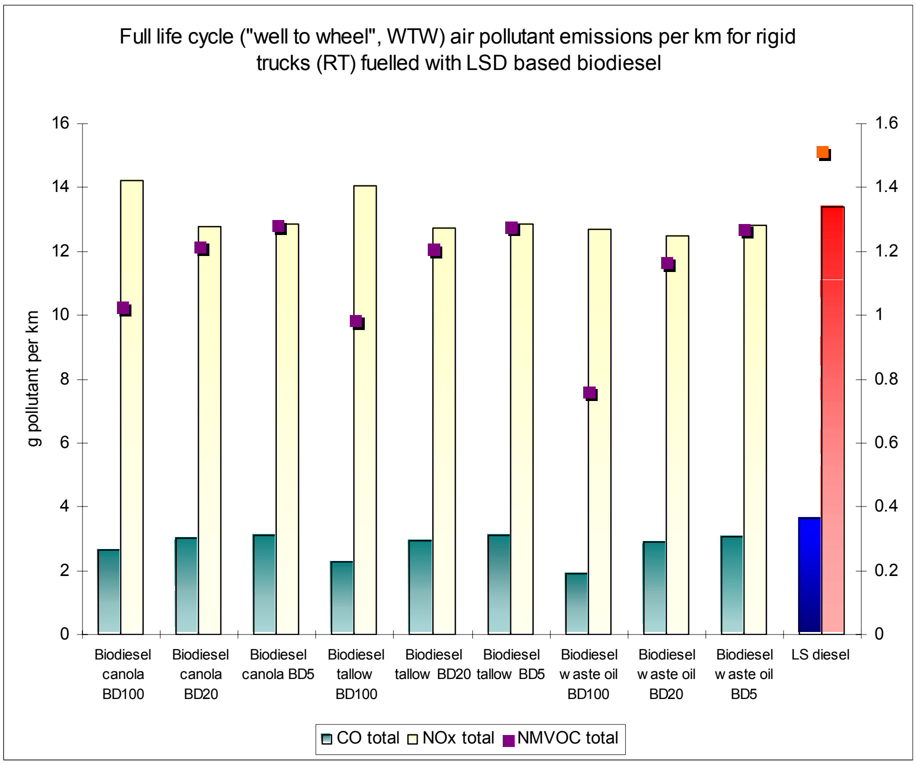


Figure A23. Fuel life cycle CO, NO_x, and NMVOC emissions from biodiesel BD100, BD20, and BD5, compared to LSD (rigid truck)

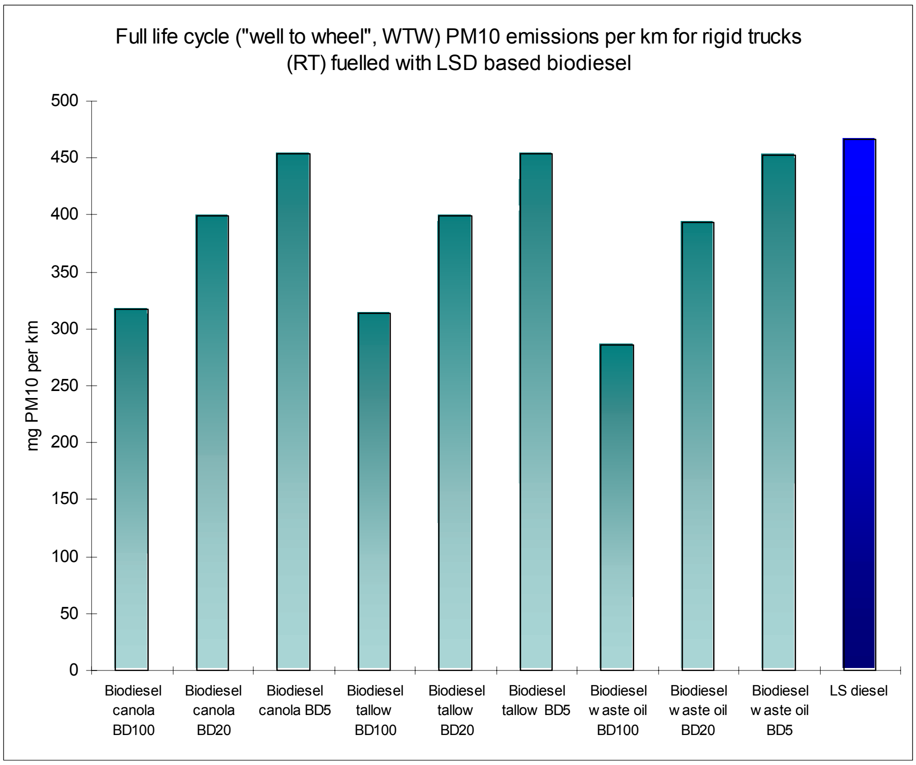


Figure A24. Fuel life cycle PM emissions from biodiesel BD100, BD20, and BD5, compared to LSD (rigid truck)

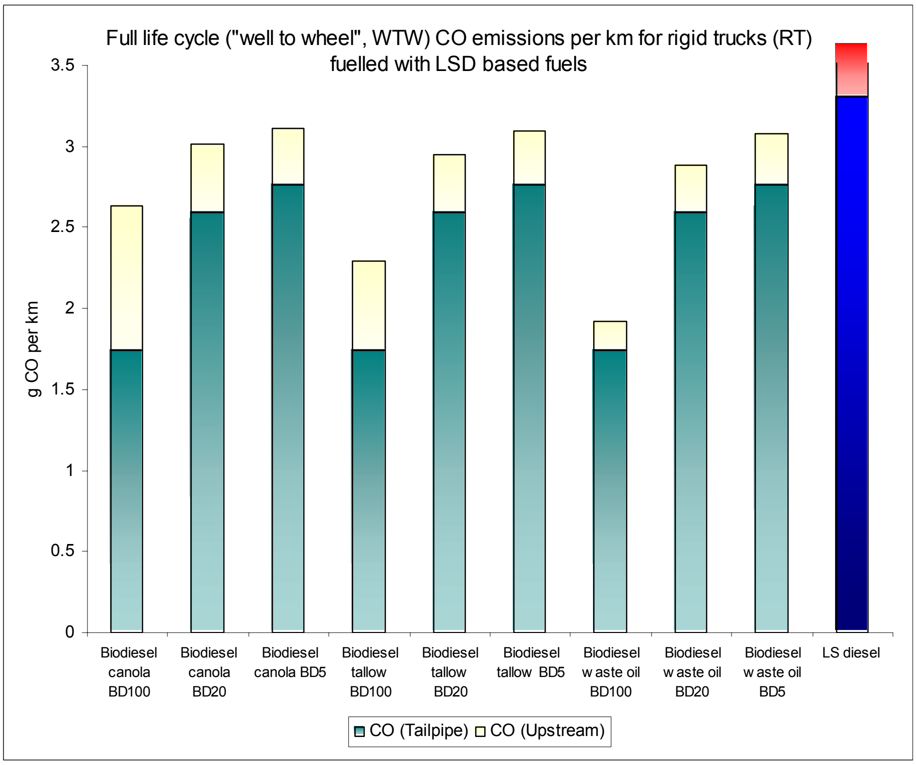


Figure A25. Fuel life cycle CO emissions from biodiesel BD100, BD20, and BD5, compared to LSD (rigid truck) separate for upstream and tailpipe

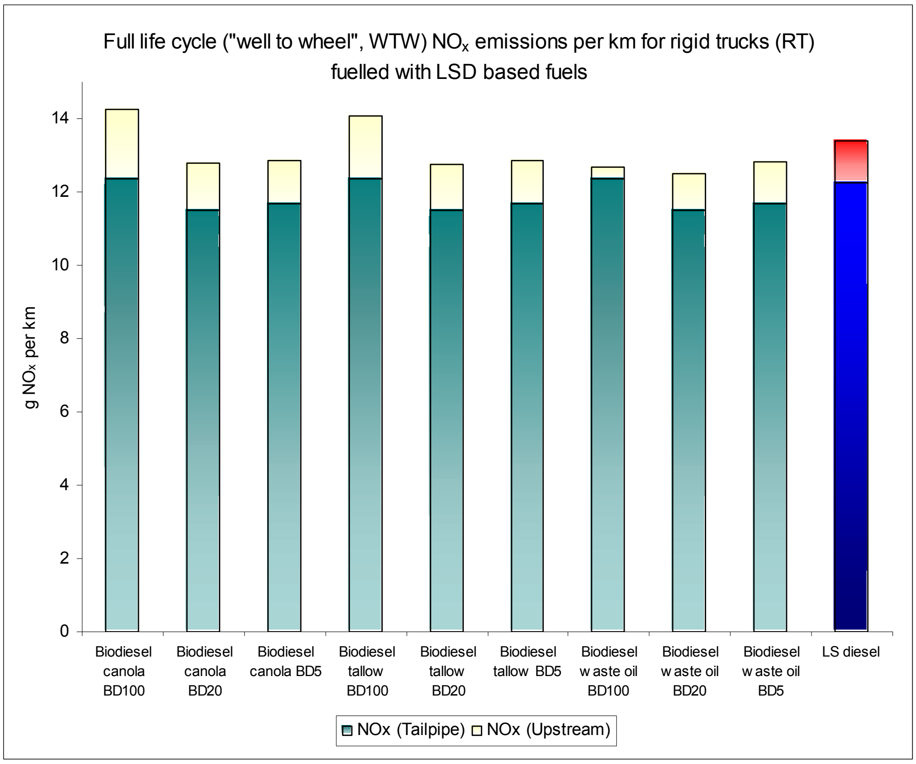


Figure A26. Fuel life cycle NO$_x$ emissions from biodiesel BD100, BD20, and BD5, compared to LSD (rigid truck) separate for upstream and tailpipe

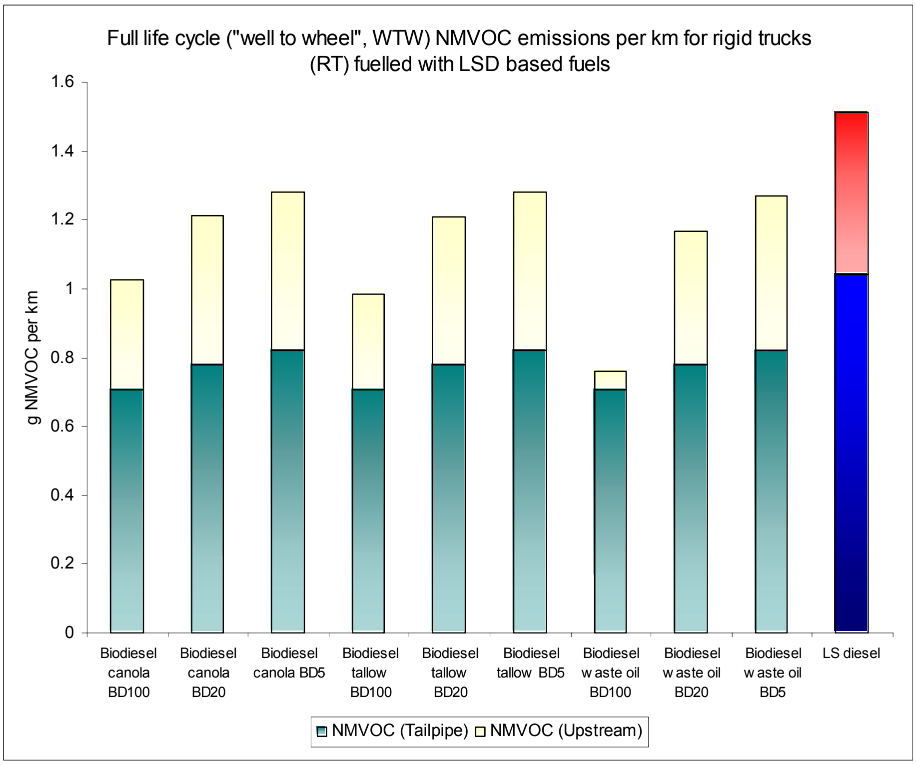


Figure A27. **Fuel life cycle NMVOC emissions from biodiesel BD100, BD20, and BD5, compared to LSD (rigid truck) separate for upstream and tailpipe**

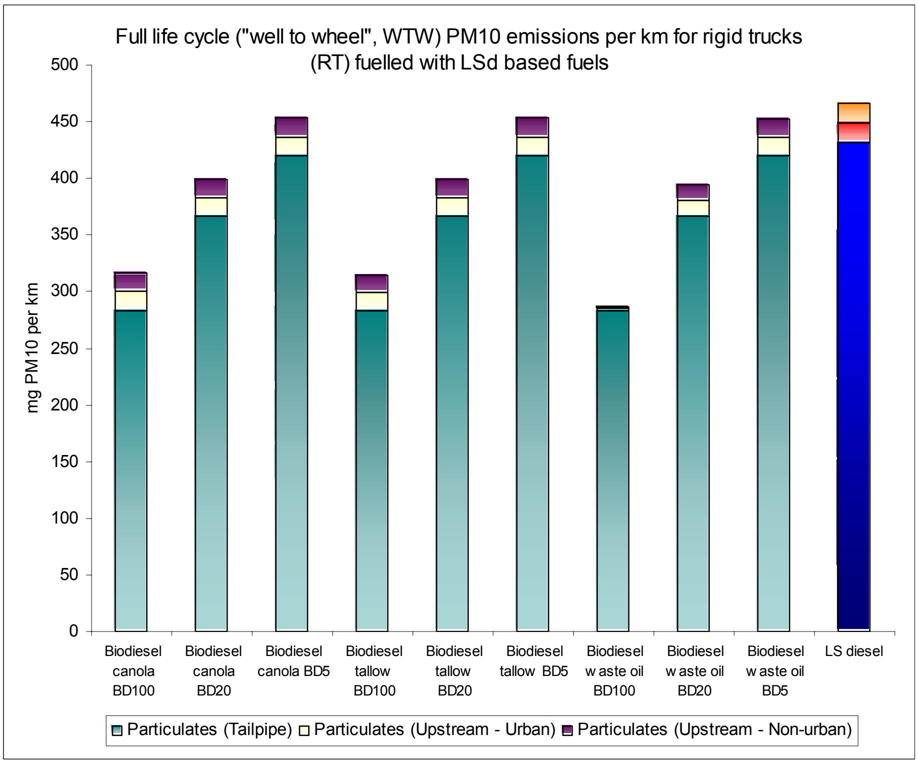


Figure A28. **Fuel life cycle PM emissions from biodiesel BD100, BD20, and BD5, compared to LSD (rigid truck) separate for upstream-urban, upstream non-urban, and tailpipe**

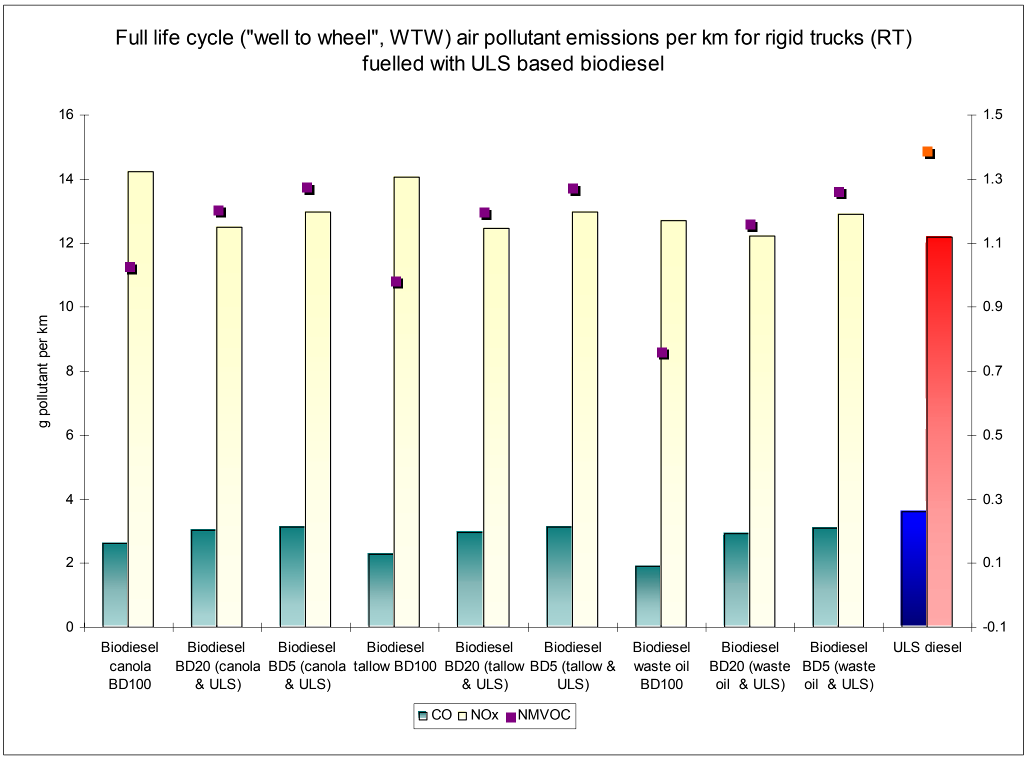


Figure A29. **Fuel life cycle CO, NO$_x$, and NMVOC emissions from biodiesel BD100, BD20, and BD5, compared to ULS (rigid truck)**

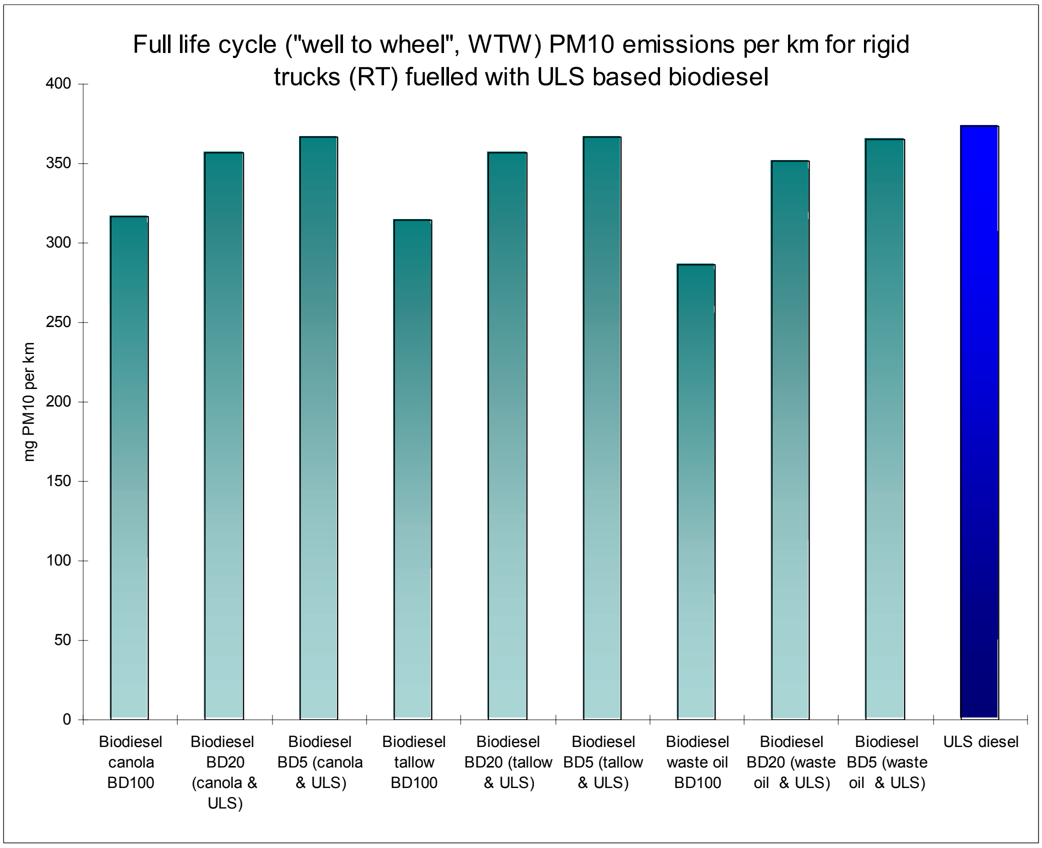


Figure A30. **Fuel life cycle PM emissions from biodiesel BD100, BD20, and BD5, compared to ULS (rigid truck)**

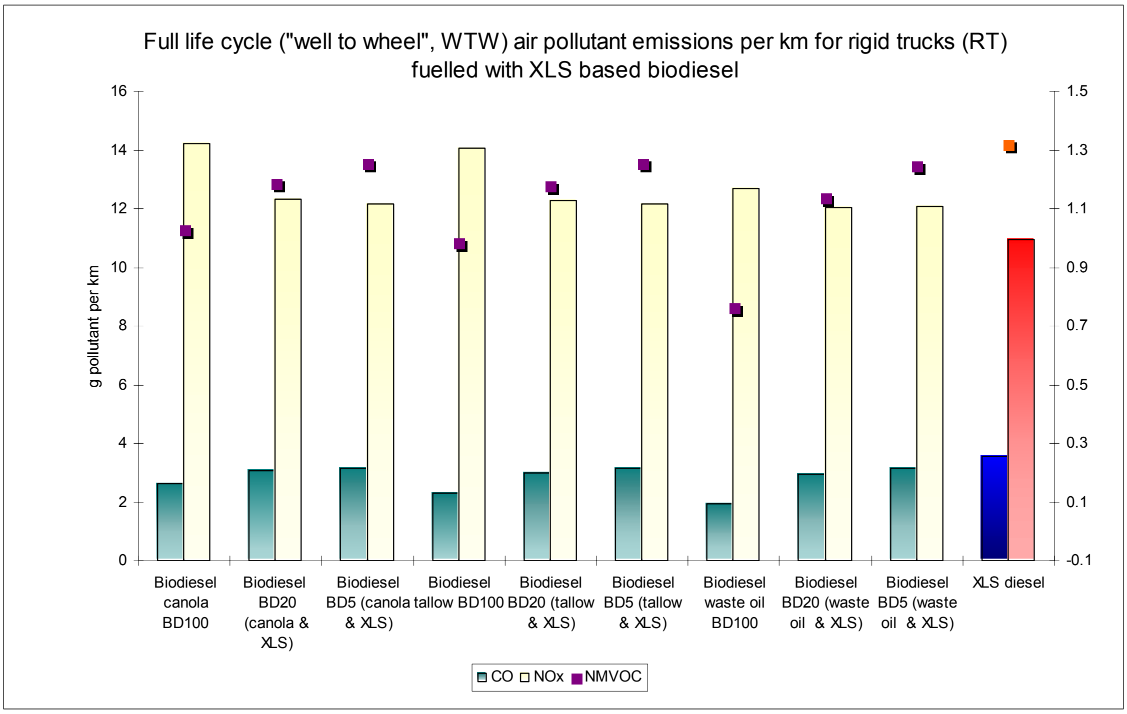


Figure A31. Fuel life cycle CO, NO_x, and NMVOC emissions from biodiesel BD100, BD20, and BD5, compared to XLS (rigid truck)

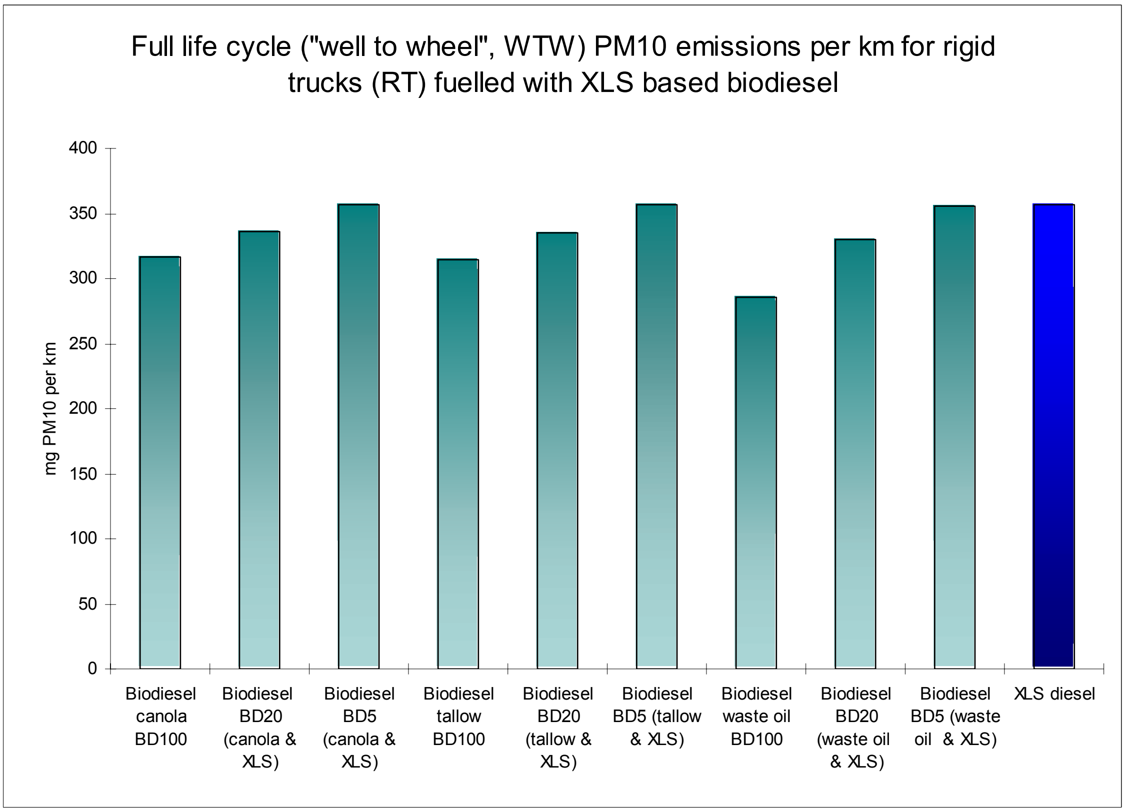


Figure A32. Fuel life cycle PM emissions from biodiesel BD100, BD20, and BD5, compared to XLS (rigid truck)

Emissions from buses, articulated trucks and 4WD

Table A67. Fuel life cycle GHG emissions from pure biodiesel and biodiesel blends compared to LSD (4WD)

Impact category	Unit	Bio diesel (canola) BD100	Bio diesel (canola) BD20	Bio diesel (canola) BD5	Bio diesel (tallow) BD100	Bio diesel (tallow) BD20	Bio diesel (tallow) BD5	Bio diesel (waste oil) BD100	Bio diesel (waste oil) BD20	Bio diesel (waste oil) BD5	**LS diesel**
CO_2 (Upstream)	g CO_2	188.4	81.66	65.69	171.2	78.63	64.94	38.52	55.27	59.12	**60.77**
Methane (Upstream)	g CH_4	0.4527	0.4958	0.5126	0.4183	0.4898	0.5111	0.06765	0.428	0.4957	**0.5215**
N_2O (Upstream)	mg N_2O	038.53	68.13	17.25	362.2	64.05	16.23	0.2	3.186	0.000344	**0.355**
CO_2 (Tailpipe)	g CO_2	0	263.7	324.1	0	263.7	324.1	0	263.7	324.1	**340.8**
Methane (Tailpipe)	g CH_4	0.0504	0.08018	0.07665	0.0504	0.08018	0.07665	0.0504	0.07018	0.07665	**0.09351**
N2O (Tailpipe)	mg N_2O	6.804	7.016	7.409	6.804	7.016	7.409	6.804	70.16	7.409	**7.461**
GHG (Upstream)	g CO_2-e	317.3497	113.1921	81.8021	292.2663	108.7713	80.7044	40.00271	64.3568	69.63646	**71.83161**
GHG (Tailpipe)	g CO_2-e	3.16764	267.5587	328.0064	3.16764	267.5587	328.0064	3.16764	286.923	328.0064	**345.0766**
GHG total	g CO_2-e	320.5173	380.7508	409.8085	295.4339	376.33	408.7108	43.17035	351.280	397.6429	**416.9082**

Table A68. Fuel life cycle GHG emissions from pure biodiesel and biodiesel blends compared to LSD (articulated trucks, AT)

Impact category	Unit	Bio diesel (canola) BD100	Bio diesel (canola) BD20	Bio diesel (canola) BD5	Bio diesel (tallow) BD100	Bio diesel (tallow) BD20	Bio diesel (tallow) BD5	Bio diesel (waste oil) BD100	Bio diesel (waste oil) BD20	Bio diesel (waste oil) BD5	**LS diesel**
CO_2 (Upstream)	g CO_2	844.1	365.8	294.4	766.9	352.2	291	172.6	247.6	264.9	**272.4**
Methane (Upstream)	g CH_4	2.028	2.221	2.297	1.874	2.194	2.291	0.3031	1.917	2.222	**2.337**
N_2O (Upstream)	mg N_2O	172.6	305.2	77.3	1623	286.	72.5	0.897	1.43	1.543	**1.592**
CO_2 (Tailpipe)	g CO_2	0	1181	1453	0	1181	1453	0	1181	1453	**1527**
Methane (Tailpipe)	g CH_4	0.3703	0.3839	0.4099	0.3703	0.3839	0.4099	0.3703	0.3839	0.4099	**0.4191**
N2O (Tailpipe)	mg N_2O	30.48	31.43	33.2	30.48	31.43	33.2	30.48	31.43	33.2	**33.43**
GHG (Upstream)	g CO_2-e	1,421.748	507.053	366.6	1,309.384	487.213	361.6635	179.2432	288.3003	312.0403	**321.9705**
GHG (Tailpipe)	g CO_2-e	17.2251	1,198.8052	1,471.8999	17.2251	1,198.805	1,471.9	17.2251	1,198.805	1,471.9	**1,546.164**
GHG total	g CO_2-e	1,438.9731	1,705.8582	1,838.4999	1,326.6091	1,686.018	1,833.563	196.4683	1,487.106	1,783.94	**1,868.135**

Table A69. Fuel life cycle GHG emissions from pure biodiesel and biodiesel blends compared to LSD (buses)

Impact category	Unit	Bio diesel (canola) BD100	Bio diesel (canola) BD20	Bio diesel (canola) BD5	Bio diesel (tallow) BD100	Bio diesel (tallow) BD20	Bio diesel (tallow) BD5	Bio diesel (waste oil) BD100	Bio diesel (waste oil) BD20	Bio diesel (waste oil) BD5	**LS diesel**
CO_2 (Upstream)	g CO_2	465.6	201.9	162.4	423	194.4	160.5	95.2	136.7	146.1	**150.2**
Methane (Upstream)	g CH_4	1.119	1.226	1.267	1.034	1.211	1.263	0.1672	1.058	1.225	**1.289**
N_2O (Upstream)	mg N_2O	952.3	168.4	42.63	895.2	158.4	40.12	0.495	0.788	0.851	**0.878**
CO_2 (Tailpipe)	g CO_2	0	652	801.1	0	652	801.1	0	652	801.1	**842.4**
Methane (Tailpipe)	g CH_4	0.2043	0.2106	0.2273	0.2043	0.2106	0.2273	0.2043	0.2106	0.2273	**0.2311**
N2O (Tailpipe)	mg N_2O	16.82	17.35	18.31	16.82	17.35	18.31	16.82	17.35	18.31	**18.44**
GHG (Upstream)	g CO_2-e	784.312	279.85	202.2223	722.226	268.935	199.4602	98.86462	159.1623	172.0889	**177.5411**
GHG (Tailpipe)	g CO_2-e	9.5045	661.8011	811.5494	9.5045	661.8011	811.5494	9.5045	661.8011	811.5494	**852.9695**
GHG total	g CO_2-e	793.8165	941.6511	1013.7717	731.7305	930.7361	1,011.01	108.3691	820.9634	983.6383	**1,030.511**

Table A70. Fuel life cycle air pollutant emissions from pure biodiesel and biodiesel blends compared to LSD (4WD)

Impact category	Unit	Bio diesel (canola) BD100	Bio diesel (canola) BD20	Bio diesel (canola) BD5	Bio diesel (tallow) BD100	Bio diesel (tallow) BD20	Bio diesel (tallow) BD5	Bio diesel (waste oil) BD100	Bio diesel (waste oil) BD20	Bio diesel (waste oil) BD5	LS diesel
CO (Tailpipe)	g CO	1.008	2.005	2.555	1.008	2.005	2.555	1.008	2.005	2.555	**2.862**
CO (Upstream)	g CO	0.3716	0.1731	0.1439	0.2288	0.148	0.1376	0.07451	0.1208	0.1308	**0.135**
NO_x (Tailpipe)	g NO_x	5.746	5.262	5.366	5.746	5.262	5.366	5.746	5.262	5.366	**5.335**
NO_x (Upstream)	g NO_x	0.7933	0.5278	0.4944	0.7189	0.5147	0.4911	0.1455	0.4137	0.466	**0.4864**
NMVOC (Tailpipe)	g NMVOC	0.2016	0.2506	0.2964	0.2016	0.2506	0.2964	0.2016	0.2506	0.2964	**0.3328**
NMVOC (Upstream)	g NMVOC	0.1338	0.1811	0.1925	0.116	0.178	0.1917	0.023	0.1616	0.1876	**0.1975**
Particulates (Tailpipe)	mg PM10	112.9	150.3	168.6	112.9	150.3	168.6	112.9	150.3	168.6	**180.4**
Particulates (Upstream-Urban)	mg PM10	7.3	7.032	7.126	6.685	6.924	7.099	1.026	5.928	6.851	**7.203**
Particulates (Upstream-Non-urban)	mg PM10	6.8	6.868	6.974	6.315	6.776	6.901	0.174	5.672	6.649	**6.997**
CO	g CO	1.3796	2.1781	2.6989	1.2368	2.153	2.6926	1.08251	2.1258	2.6858	**2.997**
NO_x	g NO_x	6.5393	5.7898	5.8604	6.4649	5.7767	5.8571	5.8915	5.6757	5.832	**5.8214**
NMVOC	g NMVOC	0.3354	0.4317	0.4889	0.3176	0.4286	0.4881	0.2246	0.4122	0.484	**0.5303**
Particulates total	mg PM10	127	164.2	182.7	125.9	164	182.6	114.1	195.6	182.1	**194.6**

Table A71. Fuel life cycle air pollutant emissions from pure biodiesel and biodiesel blends compared to LSD (articulated trucks, AT)

Impact category	Unit	Bio diesel (canola) BD100	Bio diesel (canola) BD20	Bio diesel (canola) BD5	Bio diesel (tallow) BD100	Bio diesel (tallow) BD20	Bio diesel (tallow) BD5	Bio diesel (waste oil) BD100	Bio diesel (waste oil) BD20	Bio diesel (waste oil) BD5	**LS diesel**
CO (Tailpipe)	g CO	3.261	4.856	5.169	3.261	4.856	5.169	3.261	4.856	5.169	**6.183**
CO (Upstream)	g CO	1.665	0.7757	0.6448	1.025	0.6631	0.6167	0.3338	0.5413	0.5863	**0.605**
NO_x (Tailpipe)	g NO_x	23.09	21.53	21.87	23.09	21.53	21.87	23.09	21.53	21.87	**22.9**
NO_x (Upstream)	g NO_x	3.554	2.364	2.216	3.221	2.306	2.201	0.6519	1.853	2.088	**2.18**
NMVOC (Tailpipe)	g NMVOC	1.319	1.459	1.534	1.319	1.459	1.534	1.319	1.459	1.534	**1.947**
NMVOC (Upstream)	g NMVOC	0.5997	0.8115	0.8626	0.5197	0.7974	0.8591	0.103	0.7241	0.8408	**0.885**
Particulates (Tailpipe)	mg PM10	529.5	684.9	785.2	529.5	684.9	785.2	529.5	684.9	785.2	**808.4**
Particulates (Upstream-Urban)	mg PM10	32.7	31.5	31.93	29.95	31.02	31.81	4.595	26.55	30.7	**32.28**
Particulates (Upstream-Non-urban)	mg PM10	30.4	30.4	31.07	28.25	30.08	30.99	0.805	25.25	29.8	**31.42**
CO total	g CO	4.926	5.6317	5.8138	4.286	5.5191	5.7857	3.5948	5.3973	5.7553	**6.788**
NOx total	g NO_x	26.644	23.894	24.086	26.311	23.836	24.071	23.7419	23.383	23.958	**25.08**
NMVOC total	g NMVOC	1.9187	2.2705	2.3966	1.8387	2.2564	2.3931	1.422	2.1831	2.3748	**2.832**
Particulates All	mg PM10	592.6	746.8	848.2	587.7	746	848	534.9	873.1	845.7	**872.1**

Table A72. Fuel life cycle air pollutant emissions from pure biodiesel and biodiesel blends compared to LSD (buses)

Impact category	Unit	Bio diesel (canola) BD100	Bio diesel (canola) BD20	Bio diesel (canola) BD5	Bio diesel (tallow) BD100	Bio diesel (tallow) BD20	Bio diesel (tallow) BD5	Bio diesel (waste oil) BD100	Bio diesel (waste oil) BD20	Bio diesel (waste oil) BD5	LS diesel
CO (Tailpipe)	g CO	2.491	4.956	5.684	2.491	4.956	5.684	2.491	4.956	5.684	**6.315**
CO (Upstream)	g CO	0.9184	0.4281	0.3556	0.5656	0.3659	0.3401	0.1841	0.2988	0.3234	**0.3337**
NO_x (Tailpipe)	g NO_x	14.08	13.13	13.14	14.08	13.13	13.14	14.08	13.13	13.14	**13.14**
NO_x (Upstream)	g NO_x	1.961	1.305	1.222	1.777	1.272	1.214	0.3596	1.023	1.152	**1.202**
NMVOC (Tailpipe)	g NMVOC	0.4783	0.7434	0.7831	0.4783	0.7434	0.7831	0.4783	0.7434	0.7831	**0.8083**
NMVOC (Upstream)	g NMVOC	0.3308	0.4479	0.4757	0.2867	0.4401	0.4738	0.05683	0.3996	0.4637	**0.4881**
Particulates (Tailpipe)	mg PM10	298.9	384.1	416.8	298.9	384.1	416.8	298.9	384.1	416.8	**445.8**
Particulates (Upstream-Urban)	mg PM10	18.04	17.39	17.61	16.52	17.12	17.55	2.535	14.65	16.93	**17.8**
Particulates (Upstream-Non-urban)	mg PM10	16.76	16.81	17.09	15.68	16.58	17.05	0.465	**13.95**	16.37	**17.4**
CO total	g CO	3.4094	5.3841	6.0396	3.0566	5.3219	6.0241	2.6751	**5.2548**	6.0074	**6.6487**
NOx total	g NO_x	16.041	14.435	14.362	15.857	14.402	14.354	14.4396	**14.153**	14.292	**14.342**
NMVOC total	g NMVOC	0.8091	1.1913	1.2588	0.765	1.1835	1.2569	0.53513	**1.143**	1.2468	**1.2964**
Particulates All	mg PM10	333.7	418.3	451.5	331.1	417.8	451.4	301.9	**482**	450.1	**481**

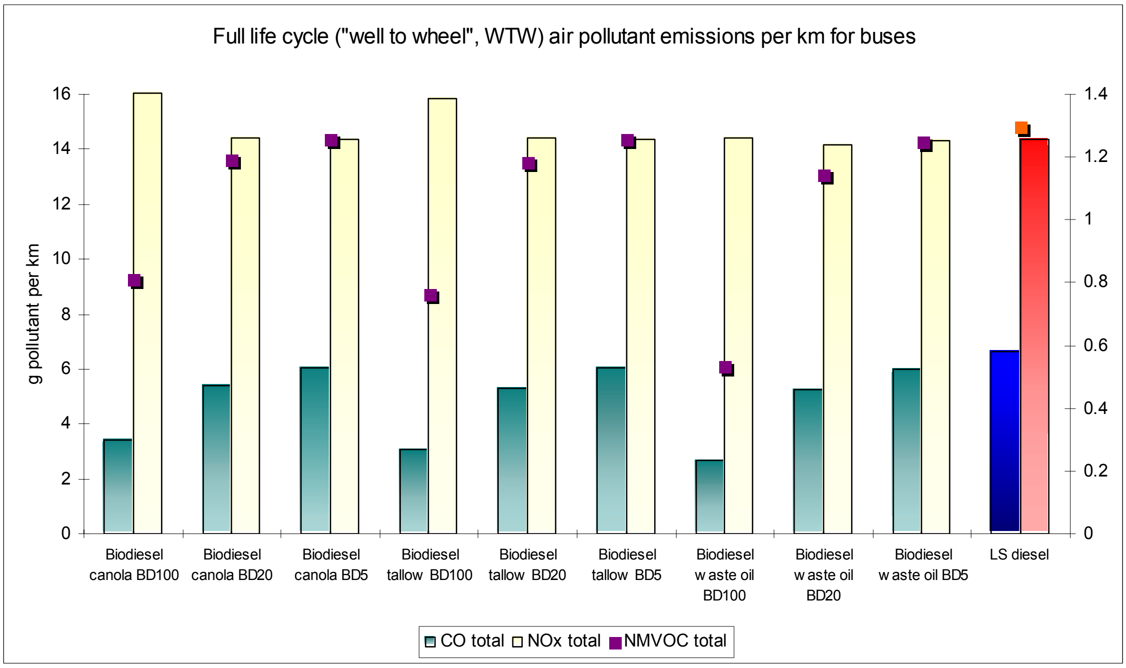


Figure A33. Fuel life cycle CO, NO$_x$, and NMVOC emissions from biodiesel BD100, BD20, and BD5, compared to LSD (buses)

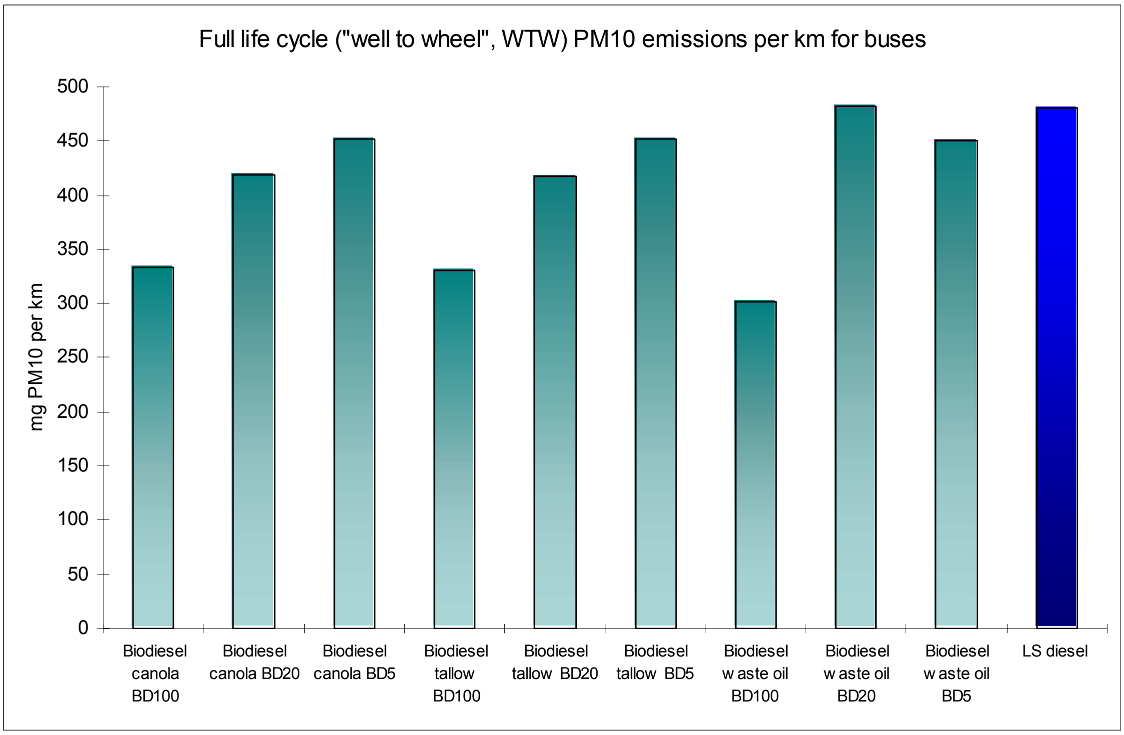


Figure A34. Fuel life cycle PM emissions from biodiesel BD100, BD20, and BD5, compared to LSD (buses)

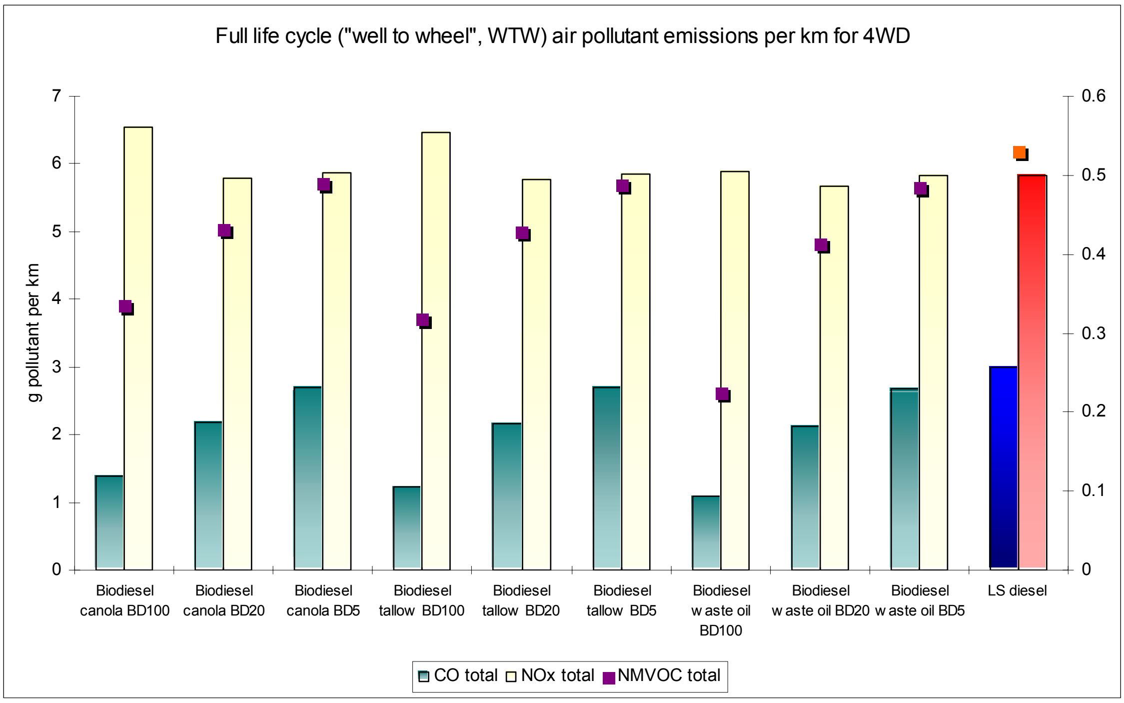


Figure A35. Fuel life cycle CO, NO_x, and NMVOC emissions from biodiesel BD100, BD20, and BD5, compared to LSD (4WD)

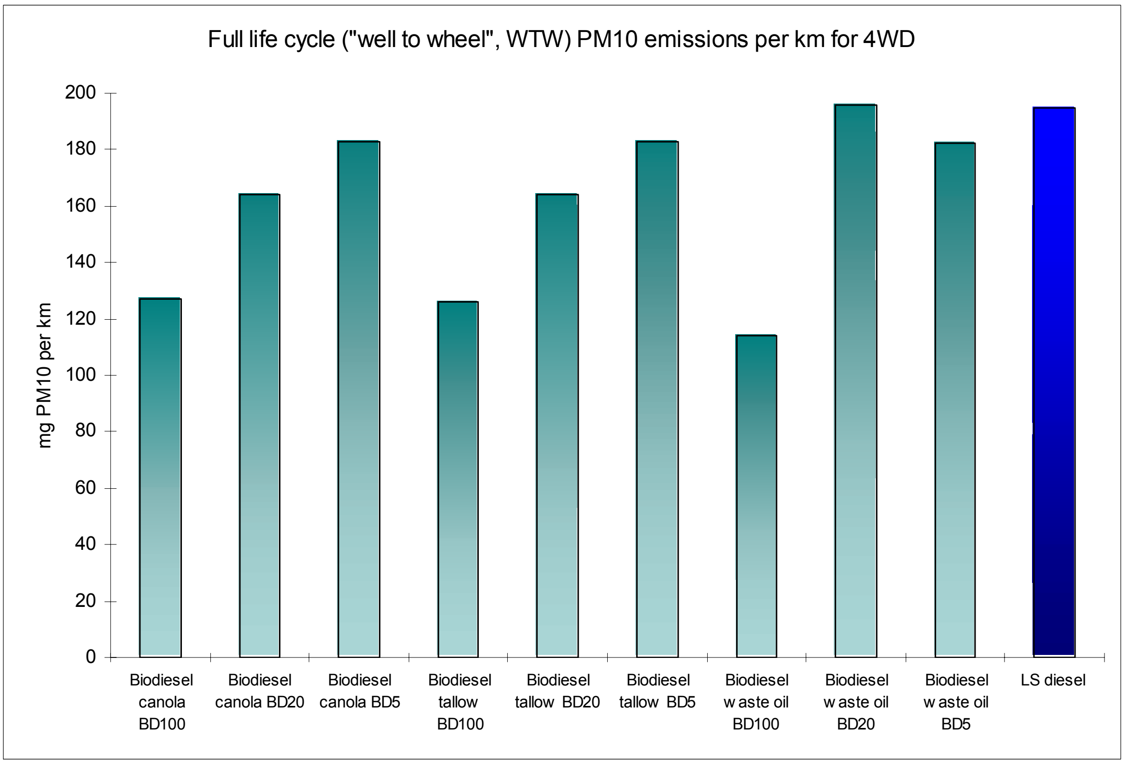


Figure A36. Fuel life cycle PM emissions from biodiesel BD100, BD20, and BD5, compared to LSD (4WD)

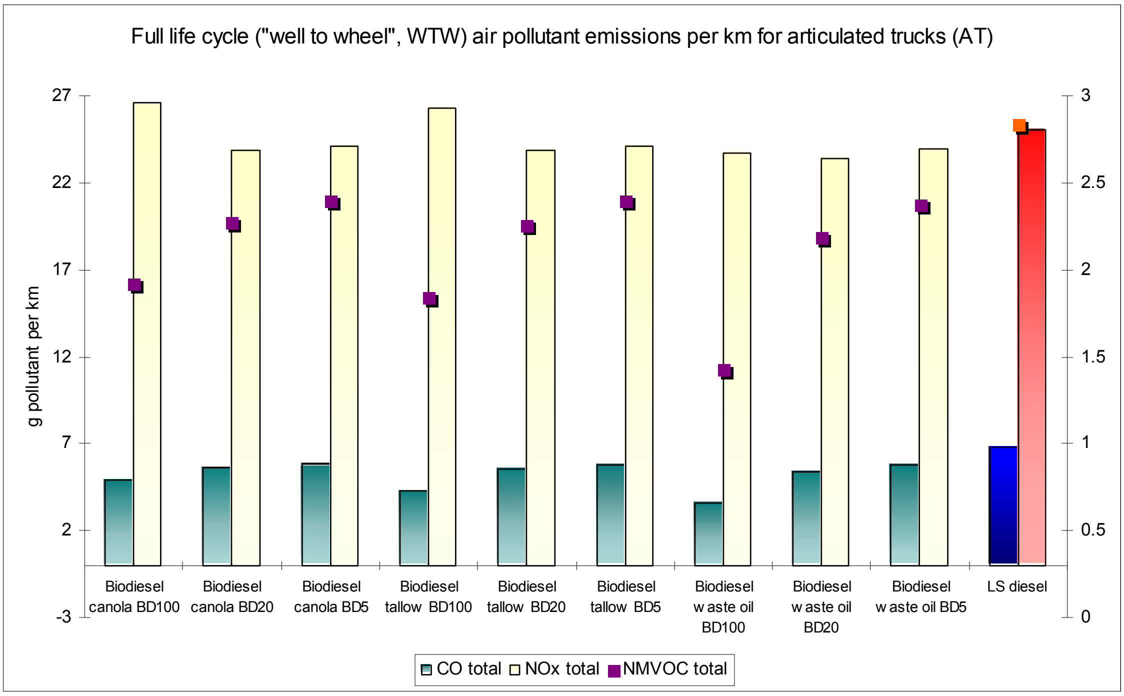


Figure A37. Fuel life cycle CO, NO_x, and NMVOC emissions from biodiesel BD100, BD20, and BD5, compared to LSD (AT)

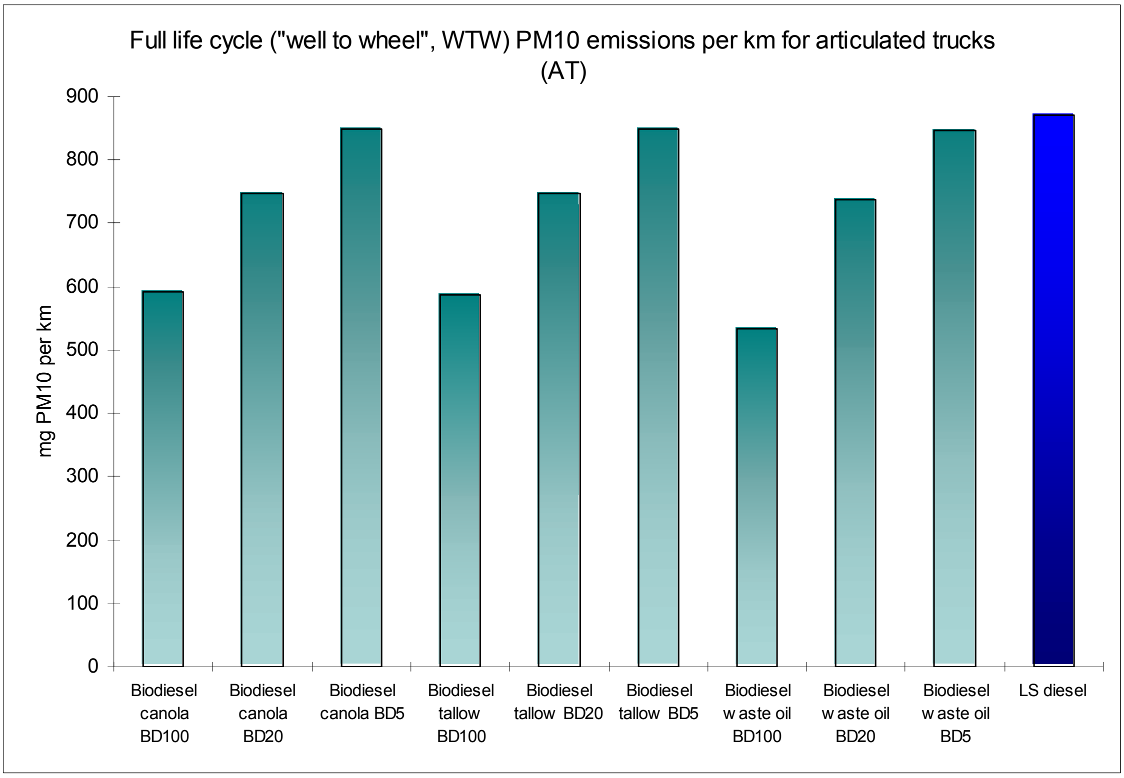


Figure A38. Fuel life cycle PM emissions from biodiesel BD100, BD20, and BD5, compared to LSD (AT)

APPENDIX VII. FUEL LIFE CYCLE PROCESS TREES

Fuel Life Cycle (WTW) Greenhouse Gas Emissions per km - ULP and Ethanol

ULP

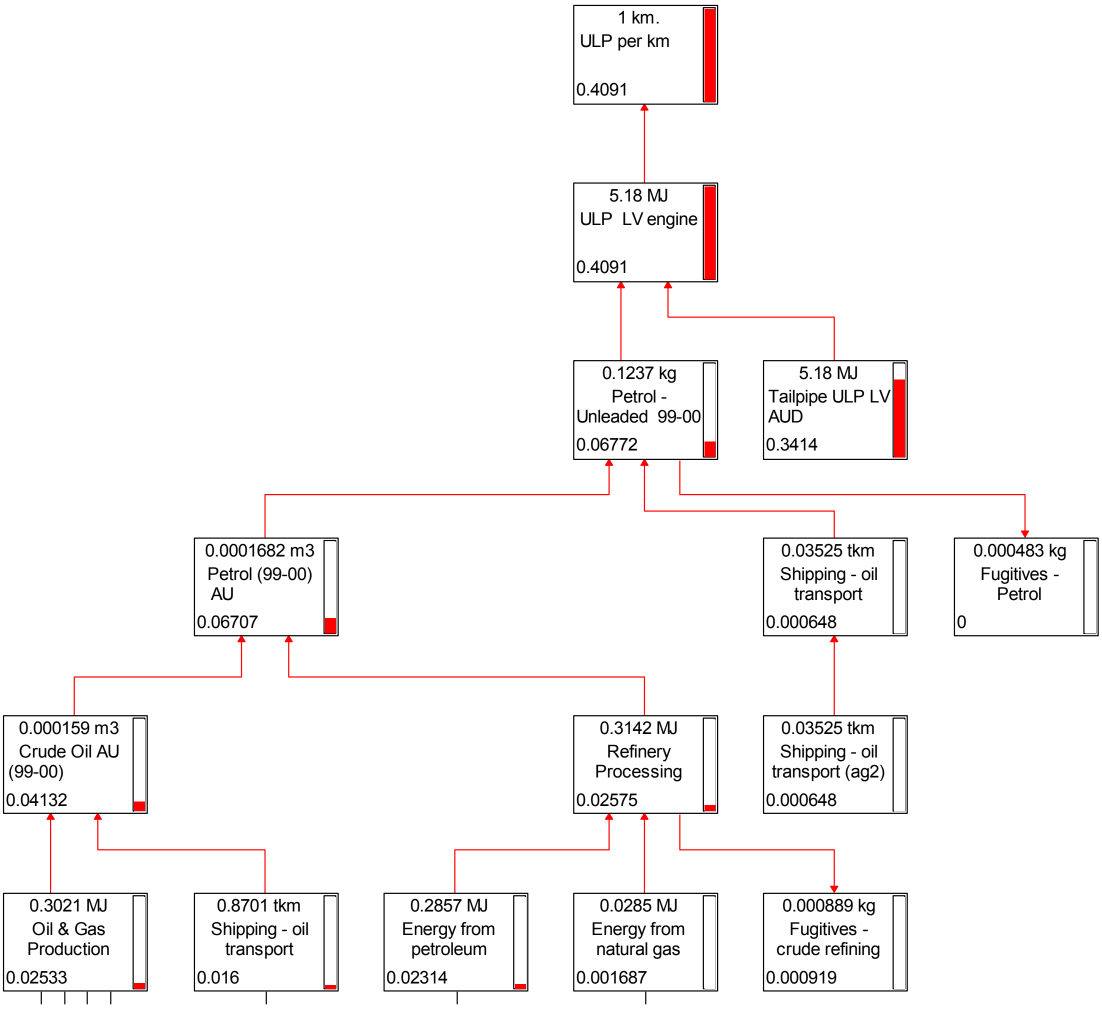


Figure A39. Unleaded petrol greenhouse emissions in kg CO_2-e per km (standardised passenger car)

Molasses

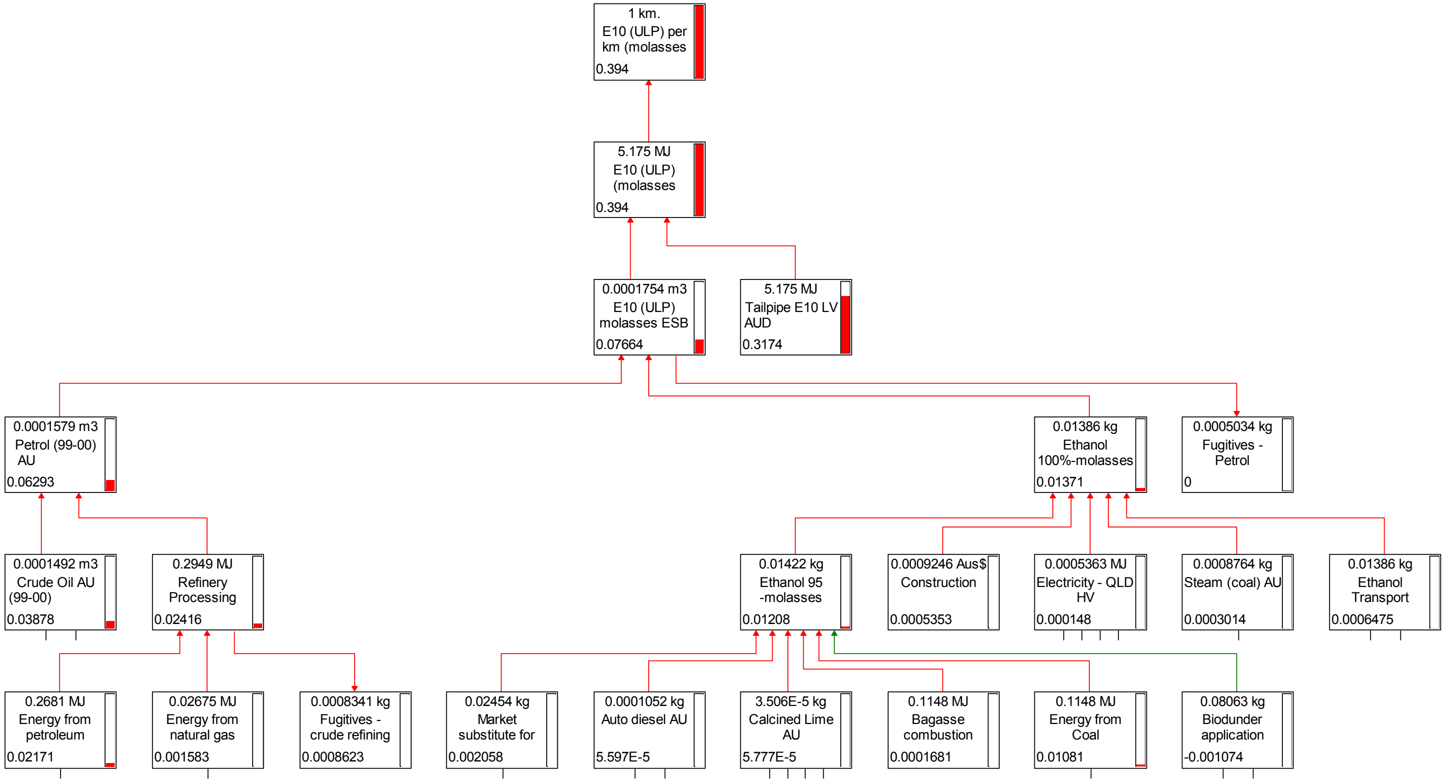


Figure A40. Ethanol (E10 - molasses) greenhouse emissions in kg CO_2-e per km (standardised passenger car)

Molasses co-generation

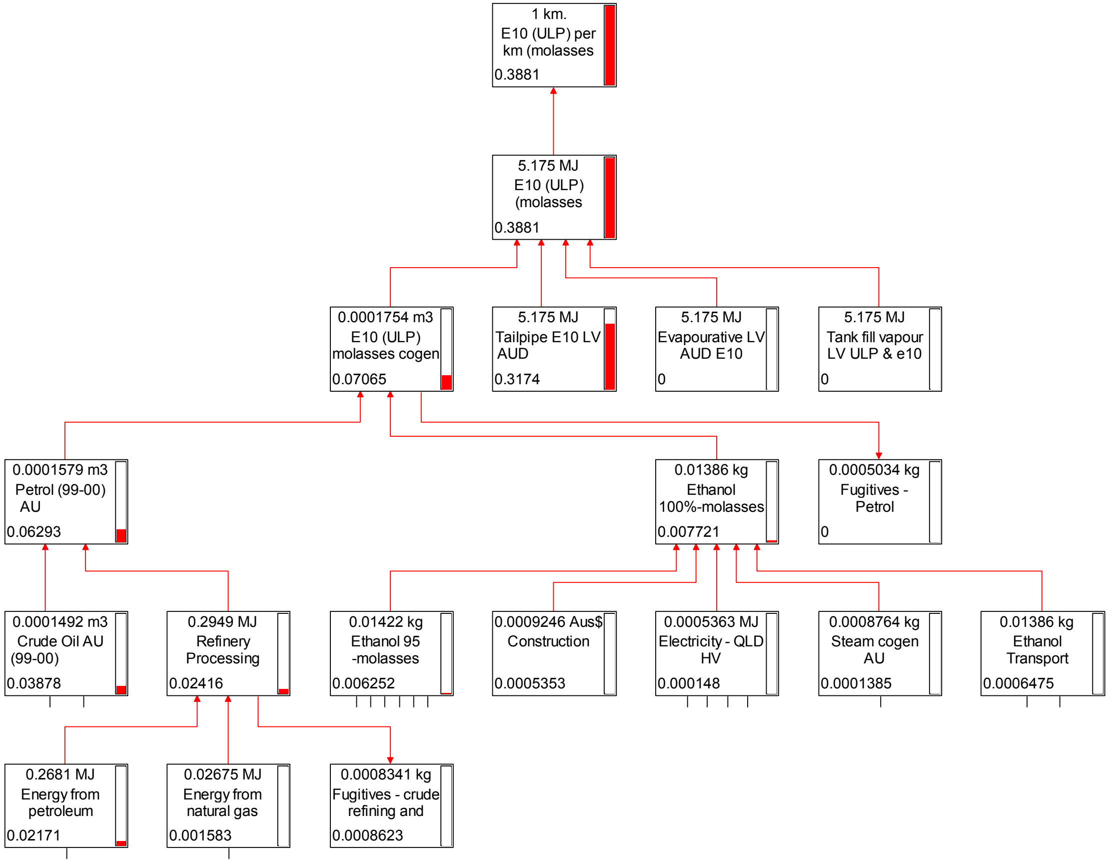


Figure A41. Ethanol (E10 – molasses co-generation) greenhouse emissions in kg CO_2-e per km (standardised passenger car)

Sorghum

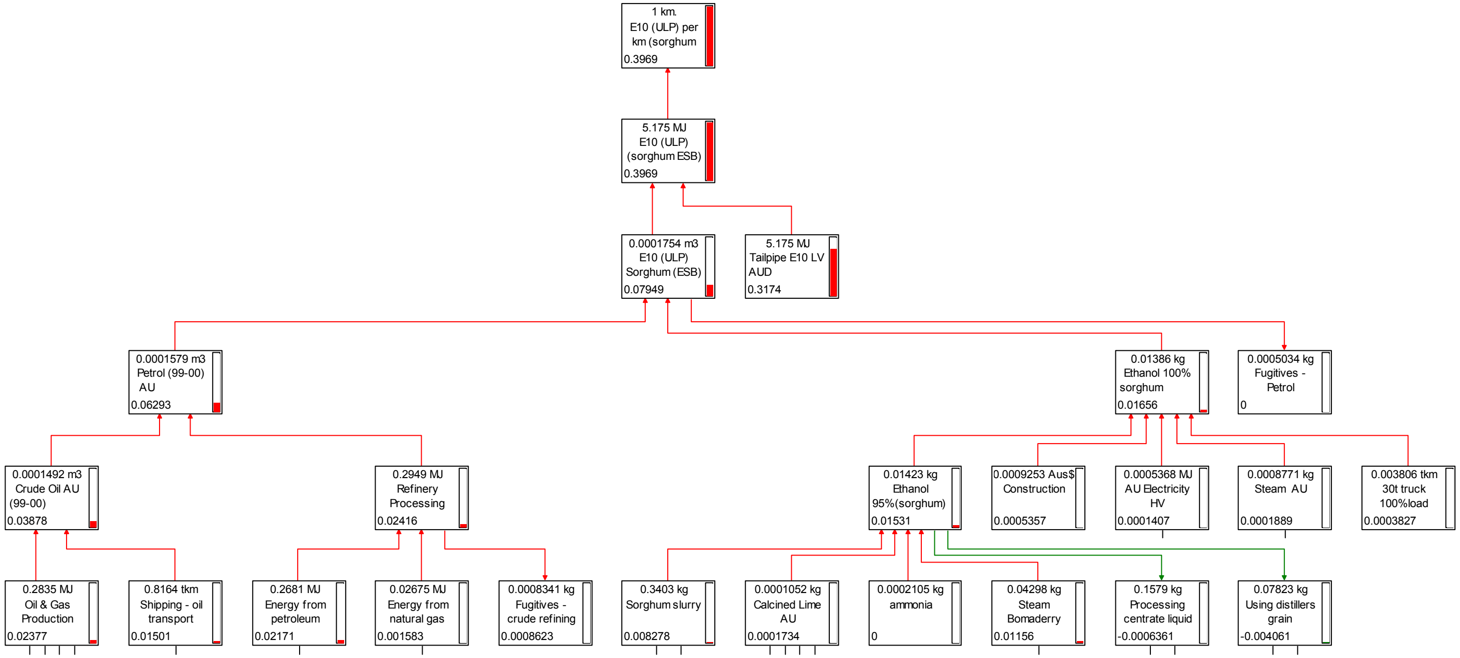


Figure A42. Ethanol (E10 - sorghum) greenhouse emissions in kg CO_2-e per km (standardised passenger car)

Wheat

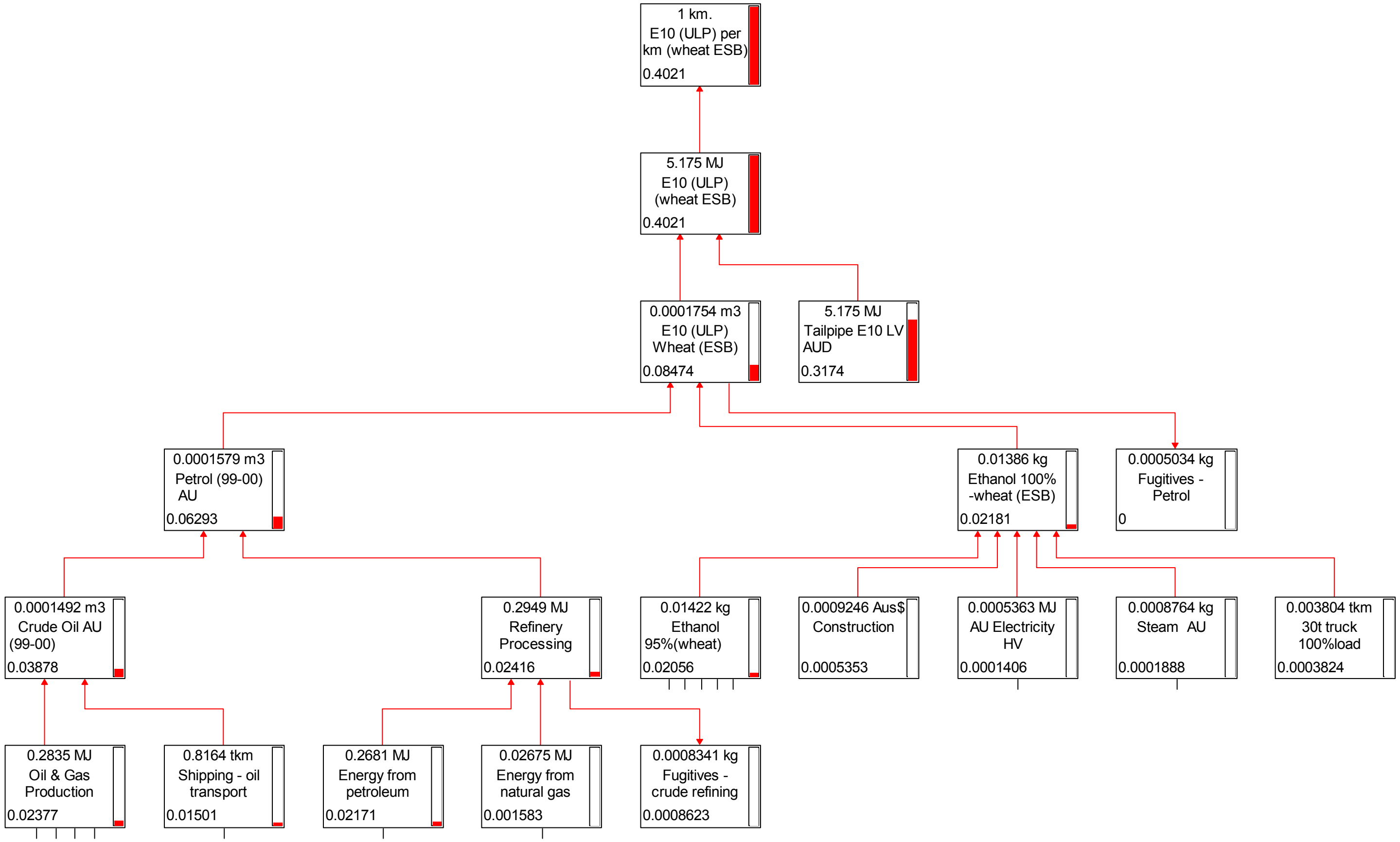


Figure A43. Ethanol (E10 - wheat) greenhouse emissions in kg CO_2-e per km (standardised passenger car)

Wheat starch waste

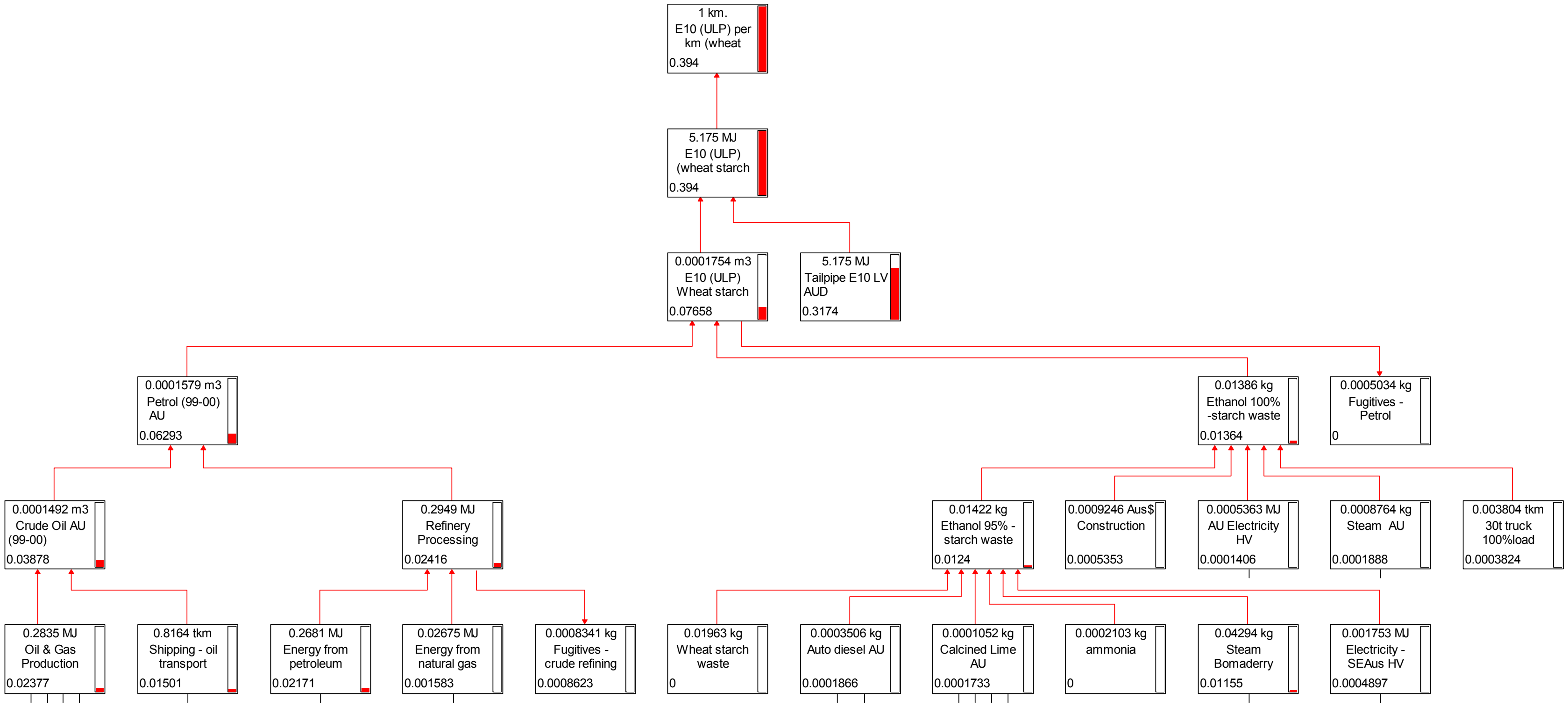


Figure A44. Ethanol (E10 – wheat starch waste) greenhouse emissions in kg CO_2-e per km (standardised passenger car)

Fuel Life Cycle (WTW) Air Pollutant Emissions per km - ULP and Ethanol

NO_x upstream emissions per km

ULP

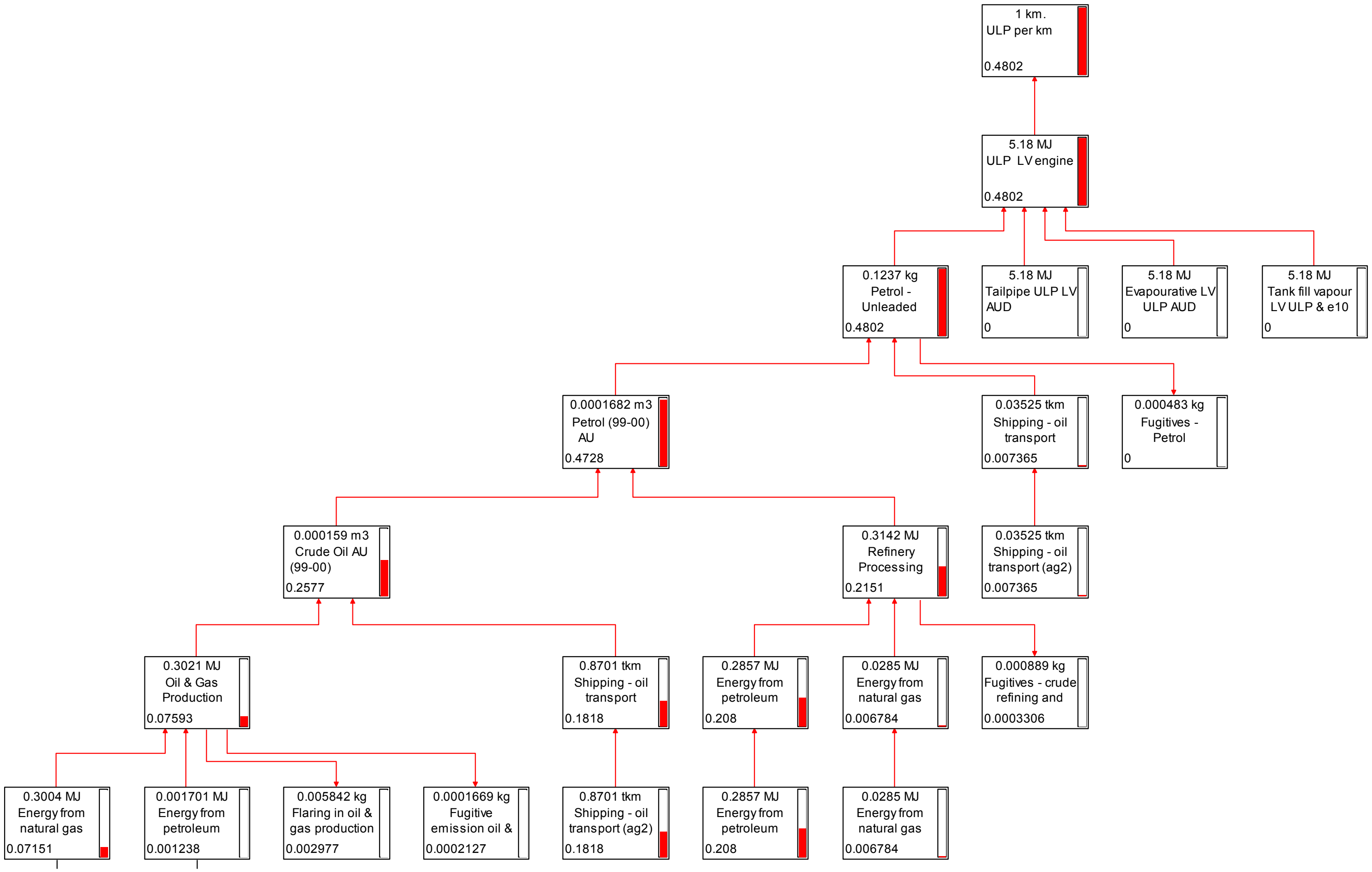


Figure A45. Unleaded petrol (ULP) emissions in g NO_x per km (standardised passenger car)

Molasses

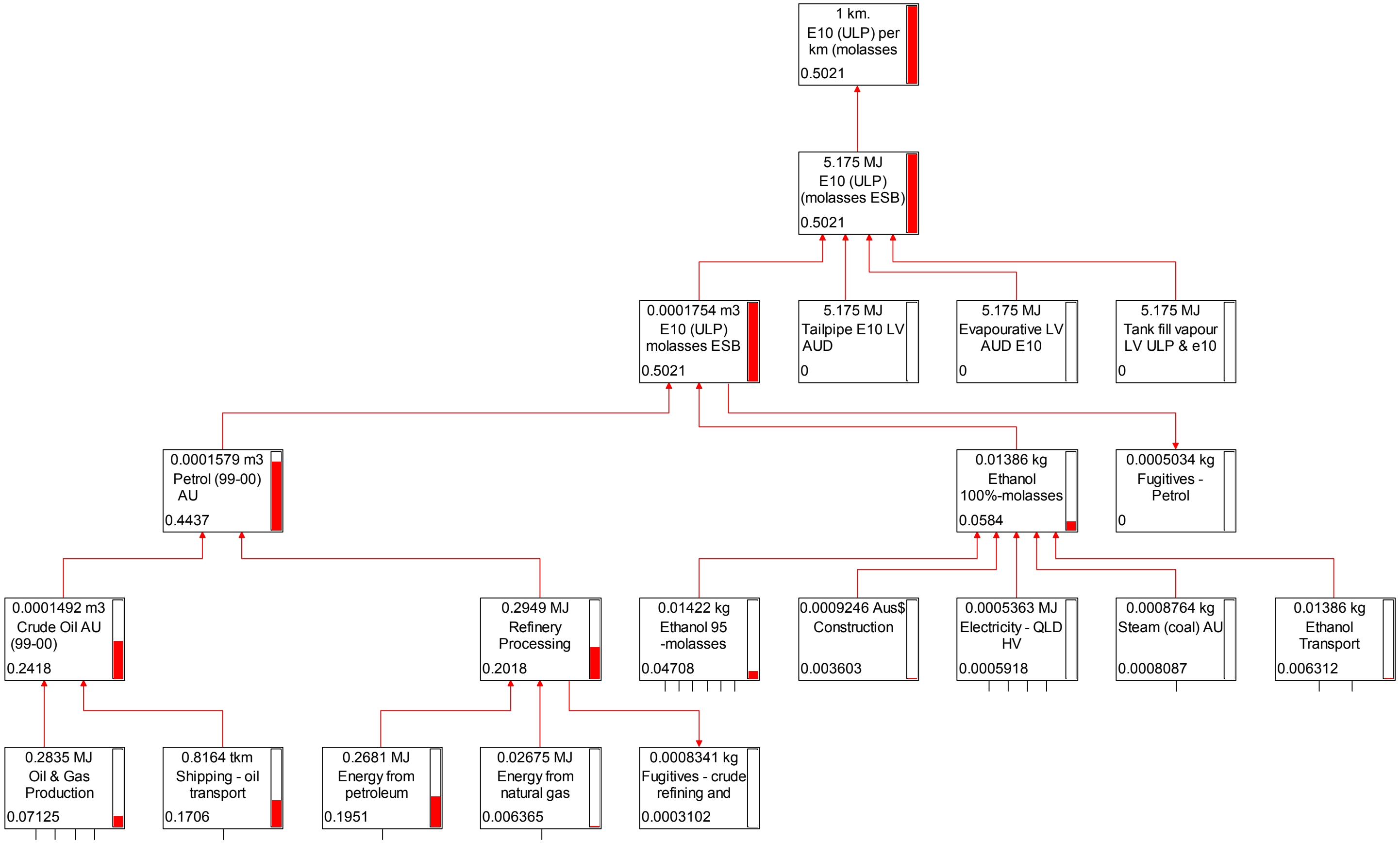


Figure A46. Ethanol (E10 - molasses) emissions in g NO_x per km (standardised passenger car)

Molasses co-generation

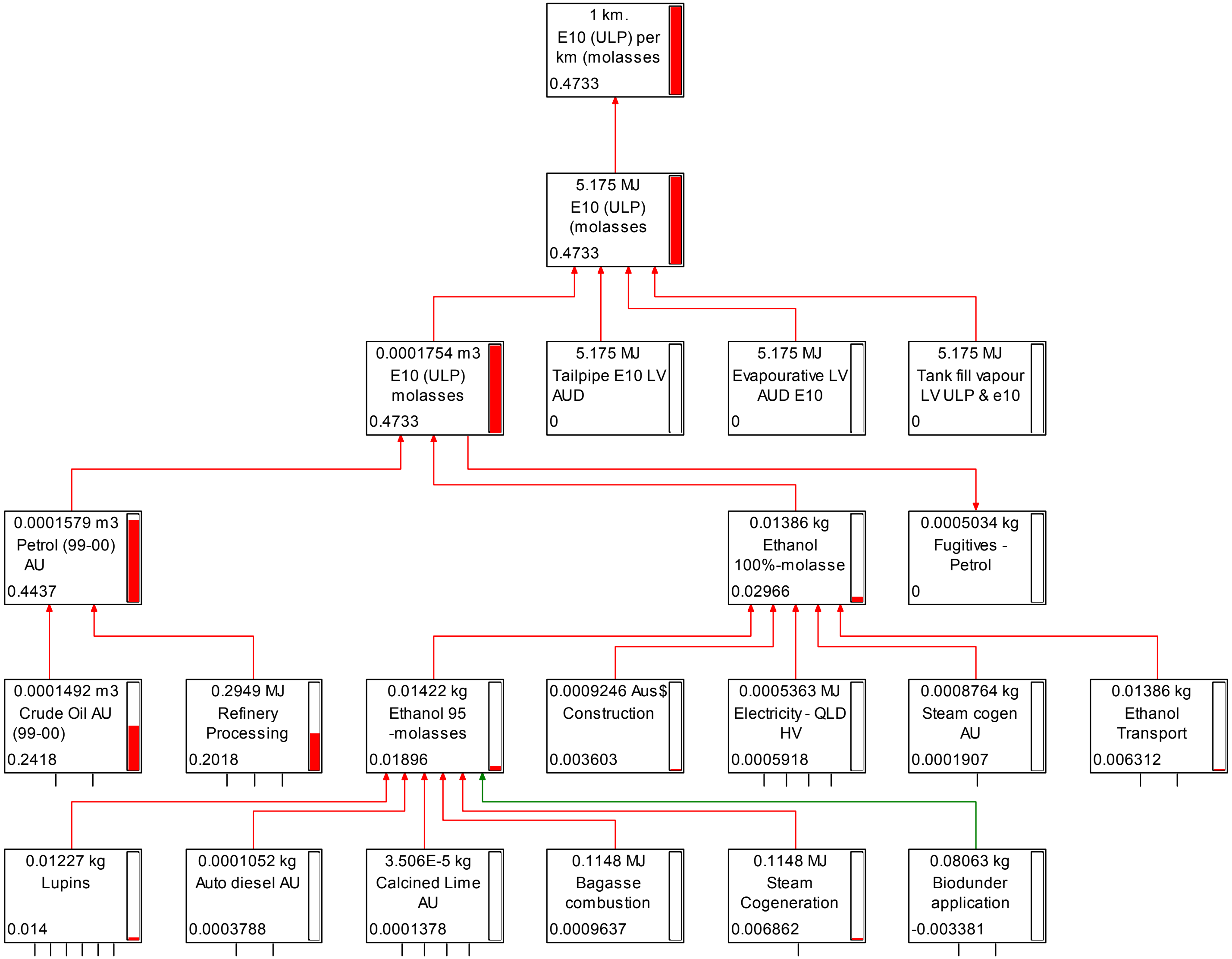


Figure A47. Ethanol (E10 – molasses co-generation) emissions in g NO_x per km (standardised passenger car)

Sorghum

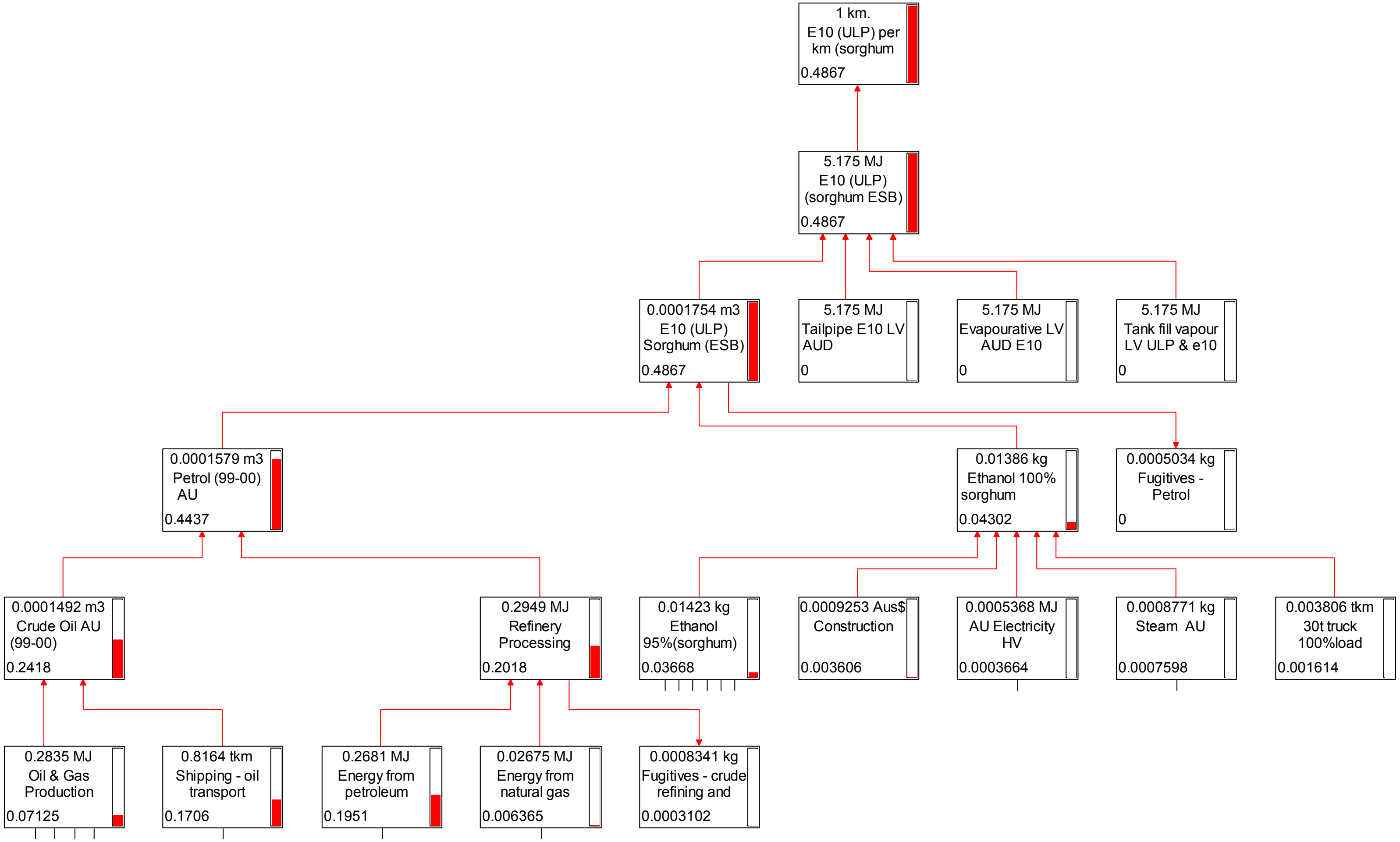


Figure A48. Ethanol (E10 - sorghum) emissions in g NO_x per km (standardised passenger car)

Wheat

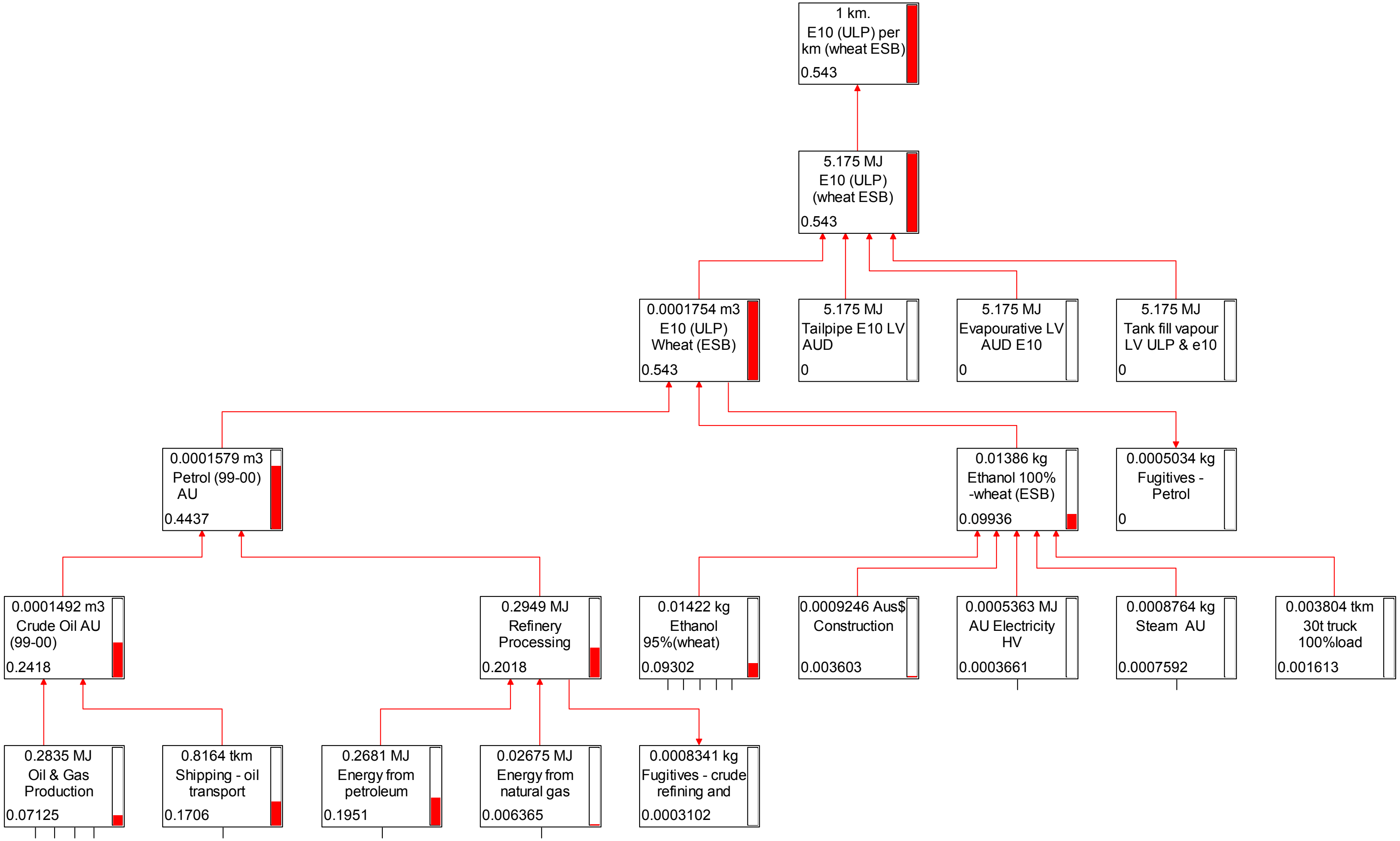


Figure A49. **Ethanol (E10 - wheat) emissions in g NO_x per km (standardised passenger car)**

Wheat starch waste

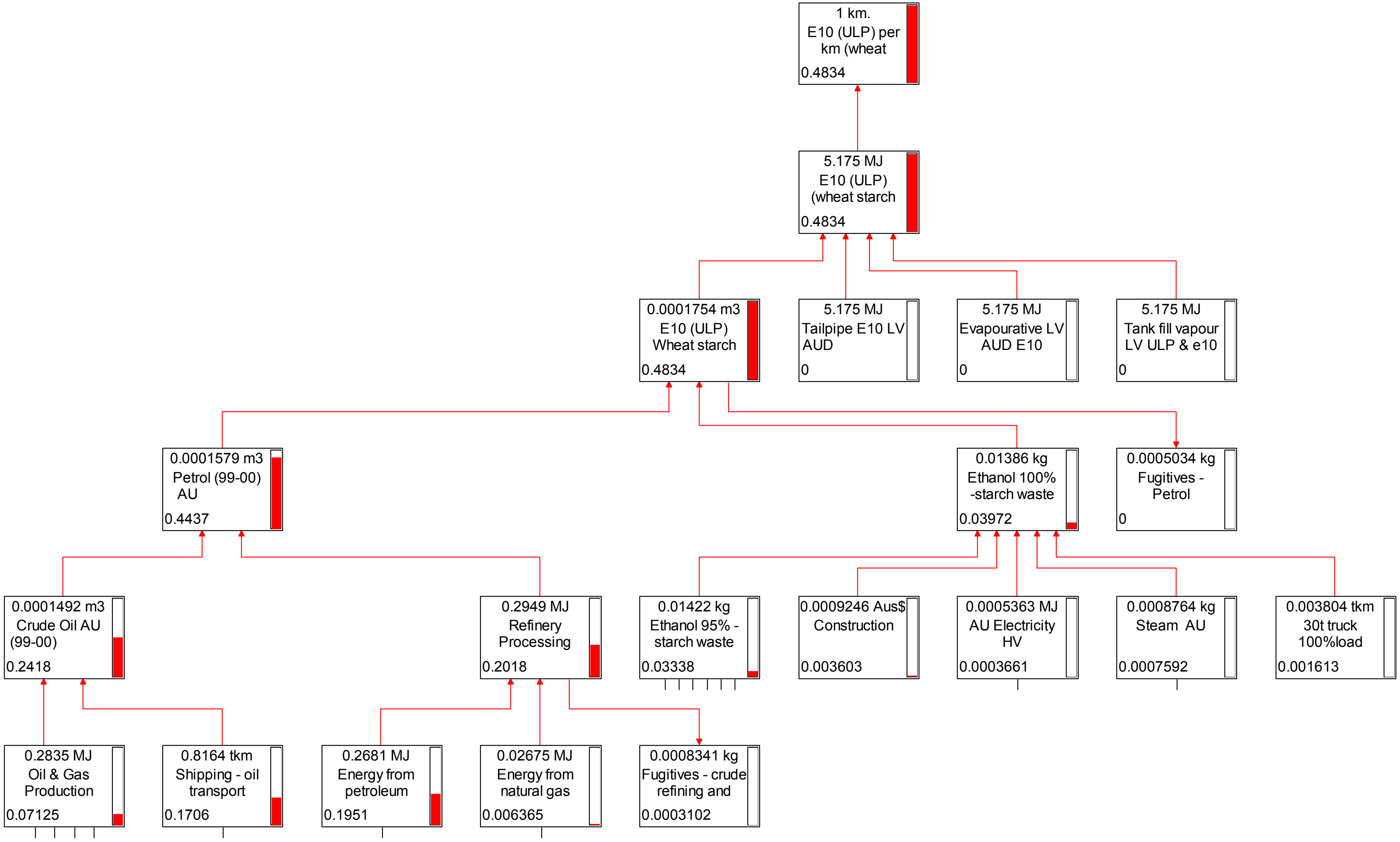


Figure A50. Ethanol (E10 – wheat starch waste) emissions in g NO_x per km (standardised passenger car)

NMHC upstream emissions per km

ULP

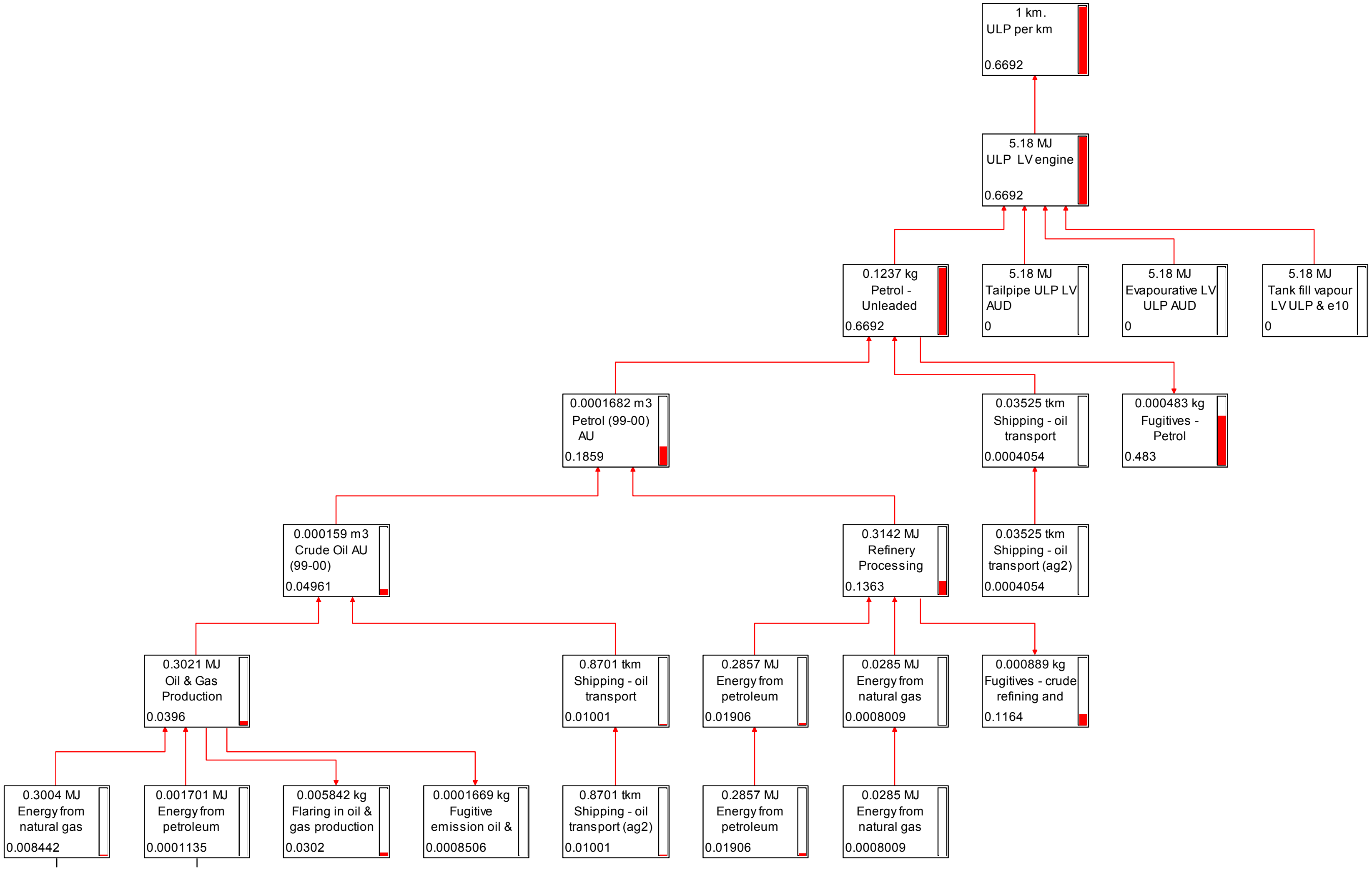


Figure A51. Unleaded petrol (ULP) emissions in g NMHC per km (standardised passenger car)

Molasses

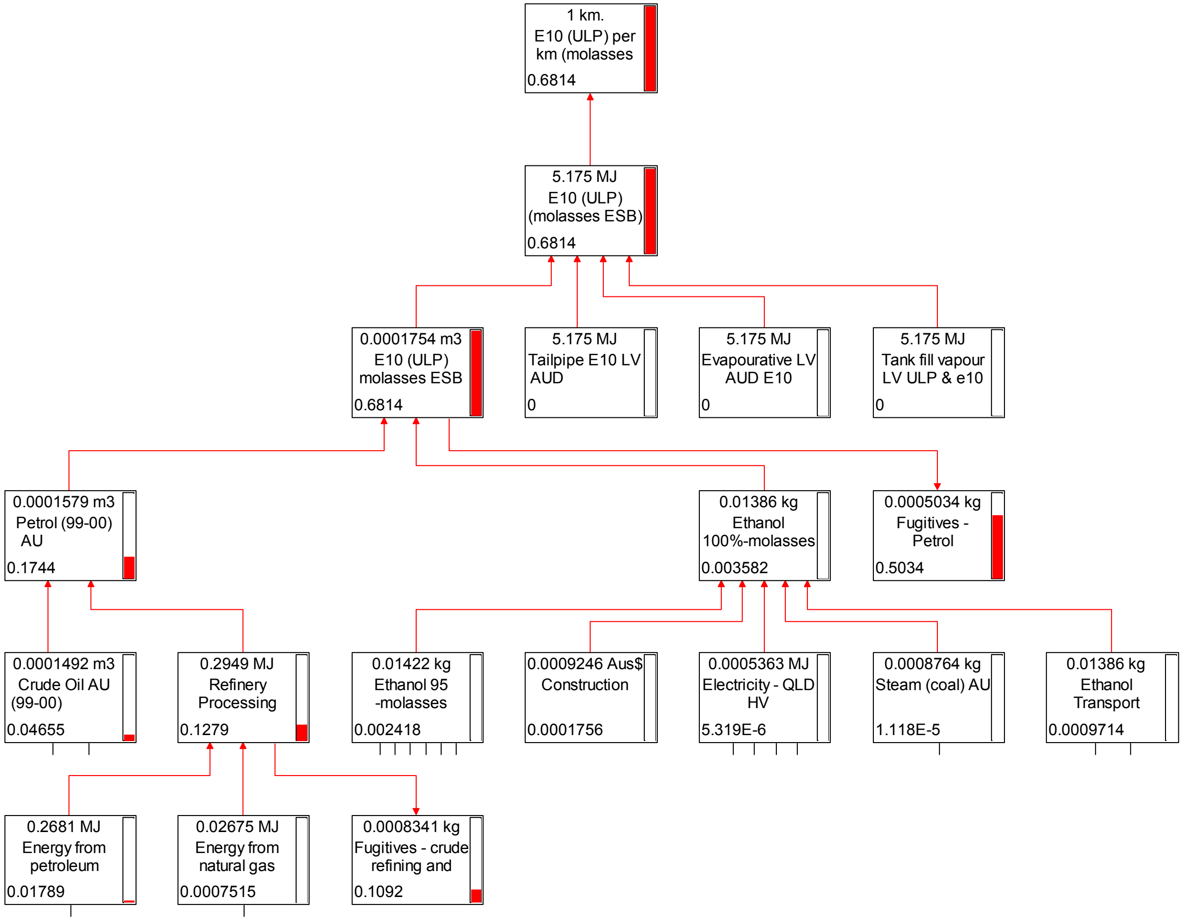


Figure A52. Ethanol (E10 - molasses) emissions in g NMHC per km (standardised passenger car)

Molasses co-generation

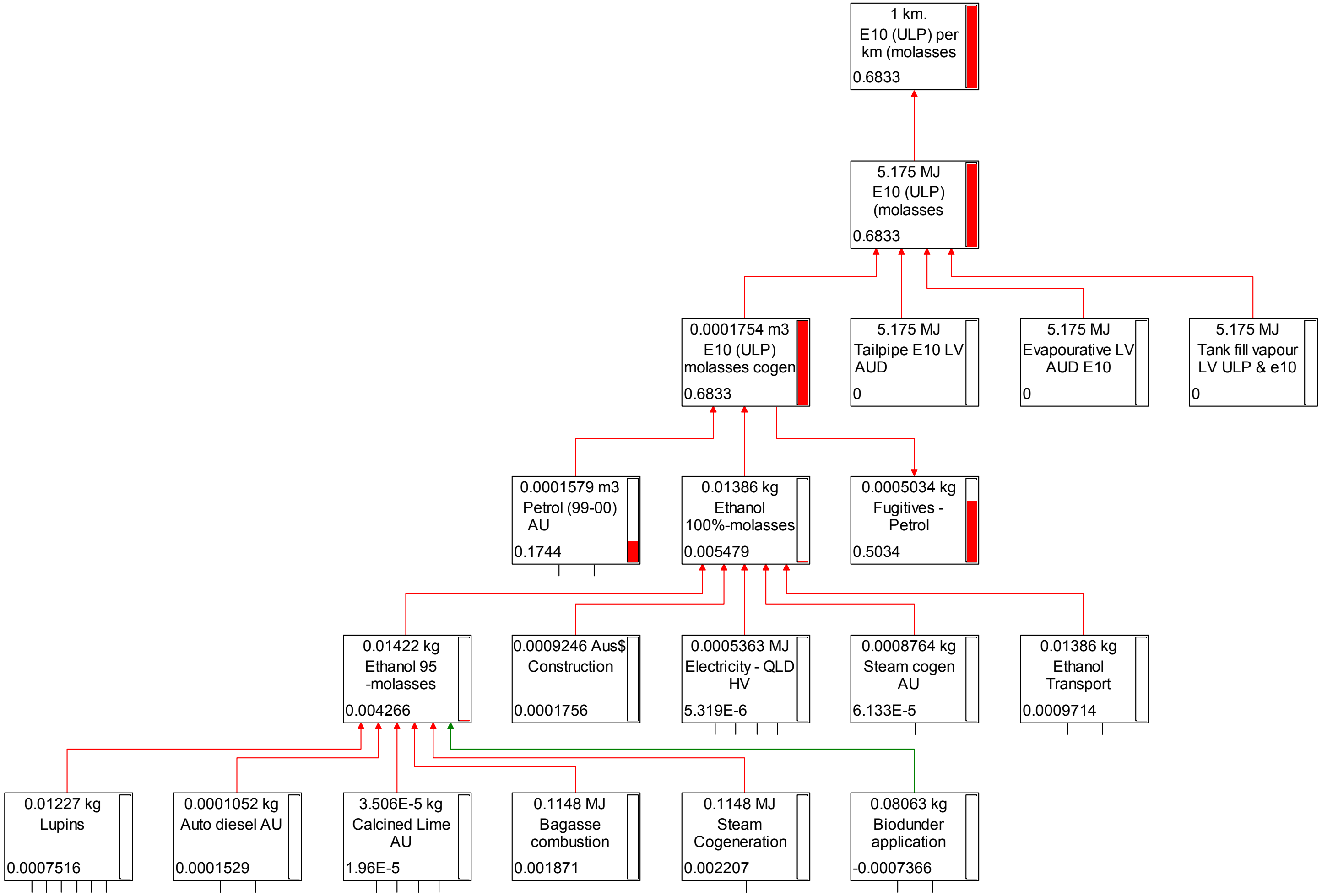


Figure A53. Ethanol (E10 - molasses co-generation) emissions in g NMHC per km (standardised passenger car)

Sorghum

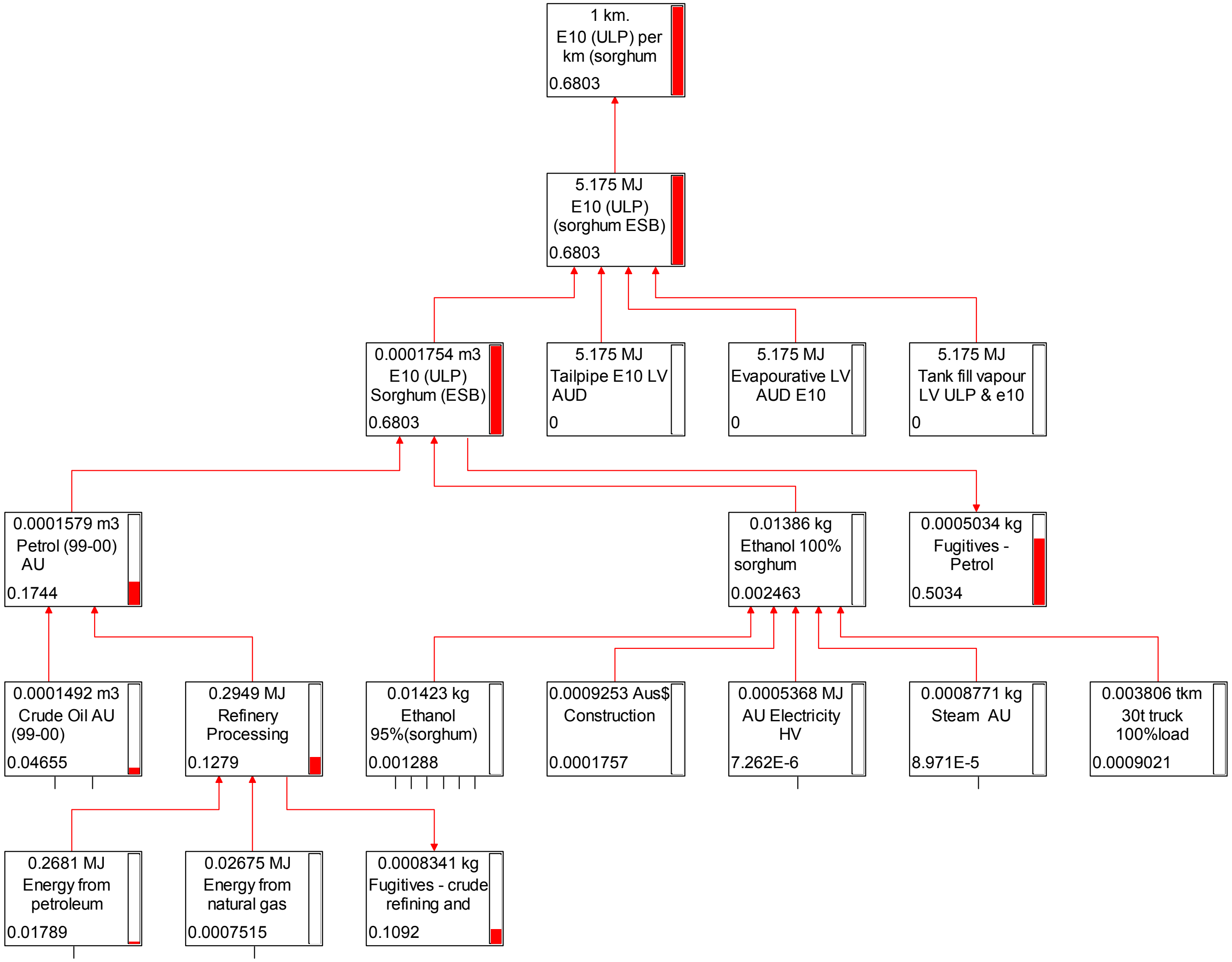


Figure A54. Ethanol (E10 - sorghum) emissions in g NMHC per km (standardised passenger car)

Wheat

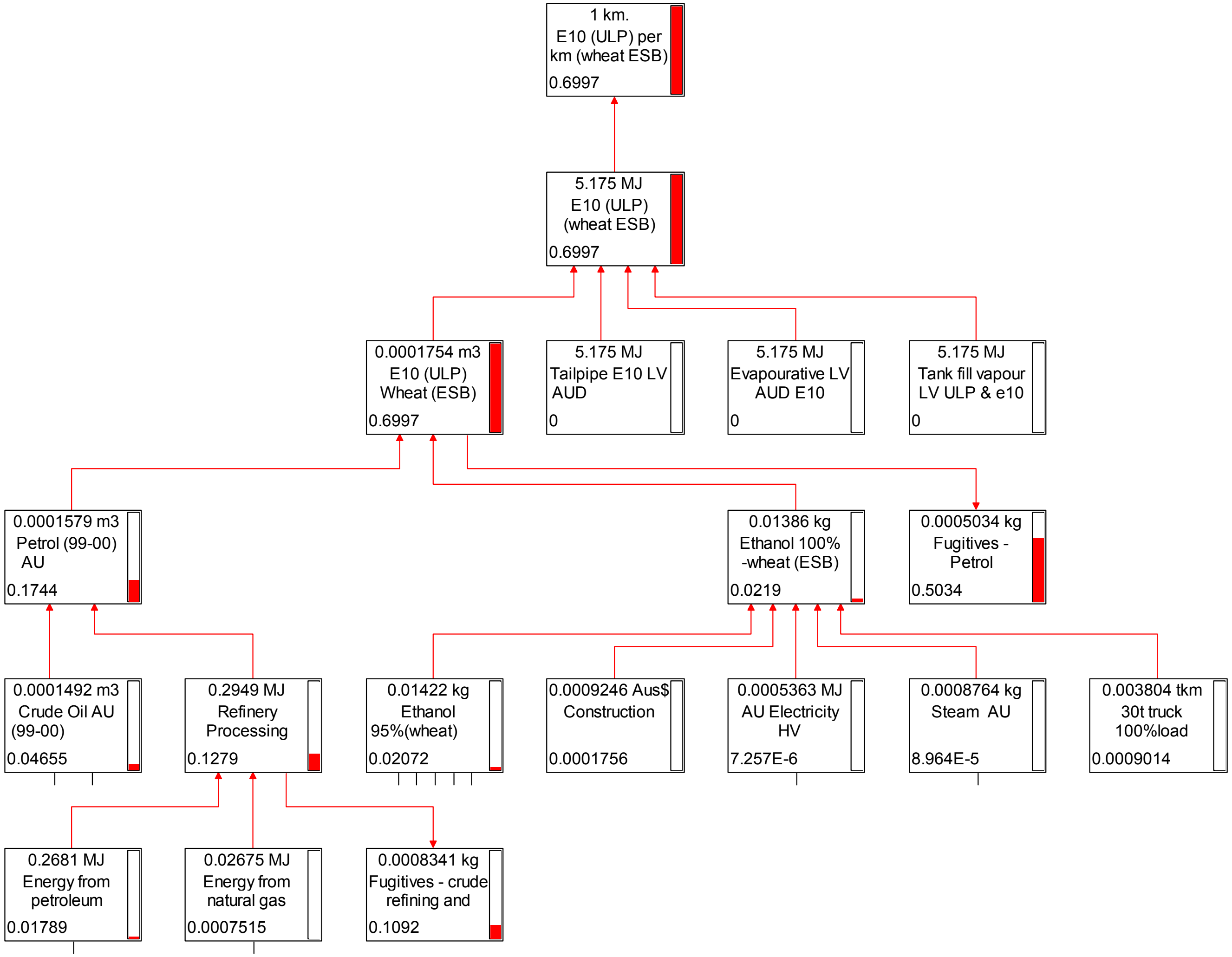


Figure A55. Ethanol (E10 - wheat) emissions in g NMHC per km (standardised passenger car)

Wheat starch waste

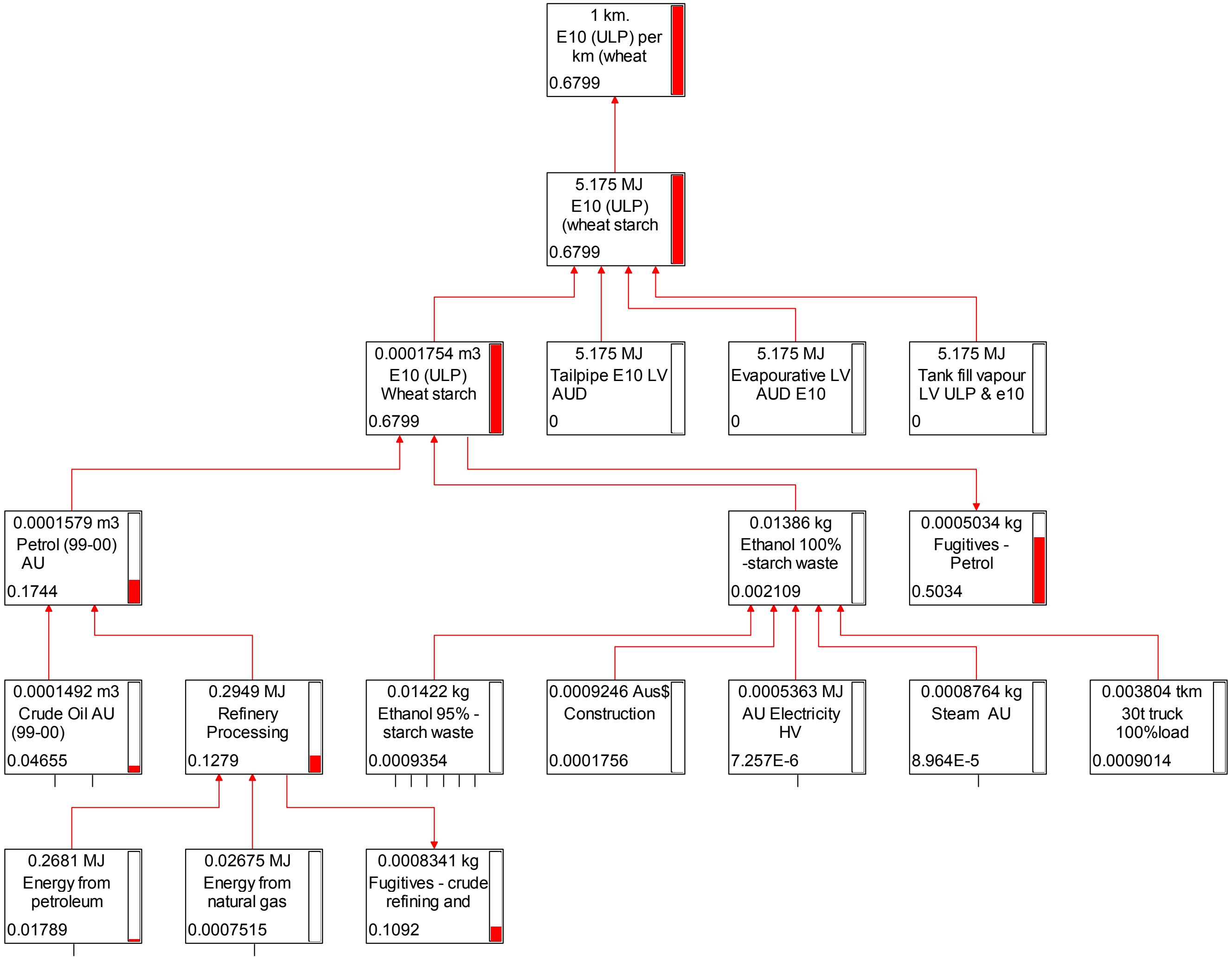


Figure A56. A2.18 Ethanol (E10 – wheat starch waste) emissions in g NMHC per km (standardised passenger car)

PM fuel life cycle (WTW) emissions per km

ULP

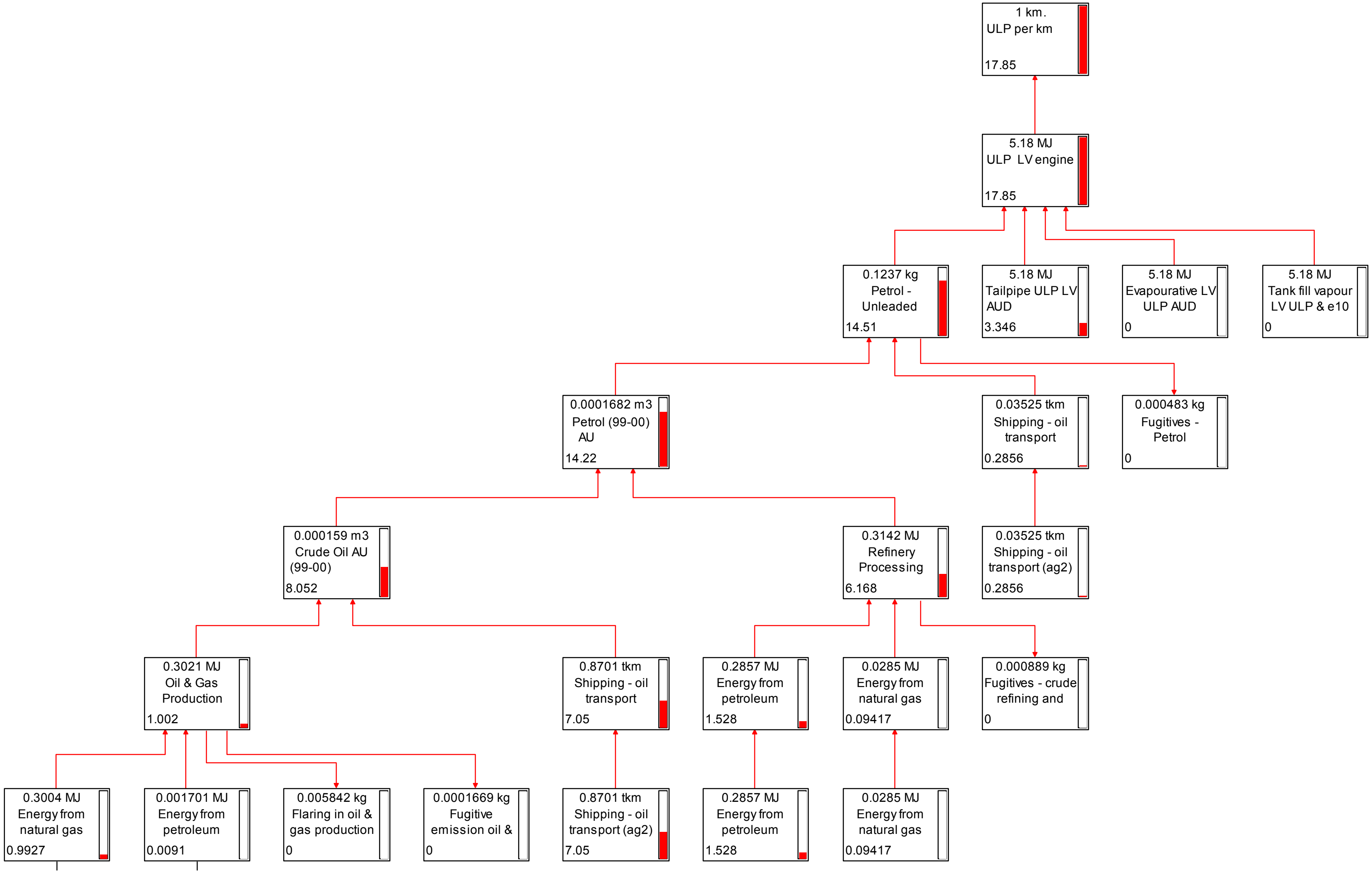


Figure A57. **Unleaded petrol (ULP) emissions in mg PM per km (standardised passenger car)**

Molasses

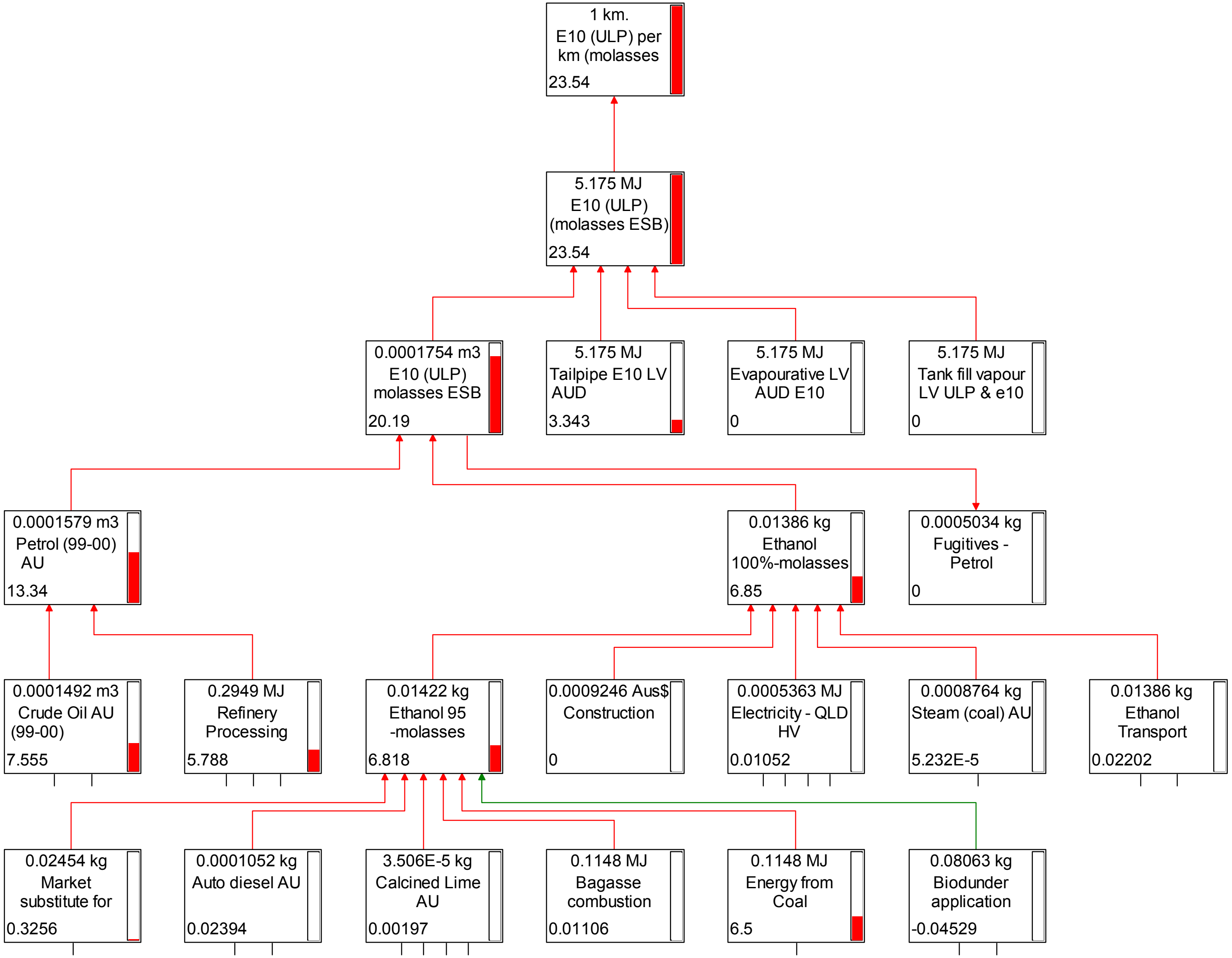


Figure A58. Ethanol (E10 - molasses) emissions in mg PM per km (standardised passenger car)

Molasses co-generation

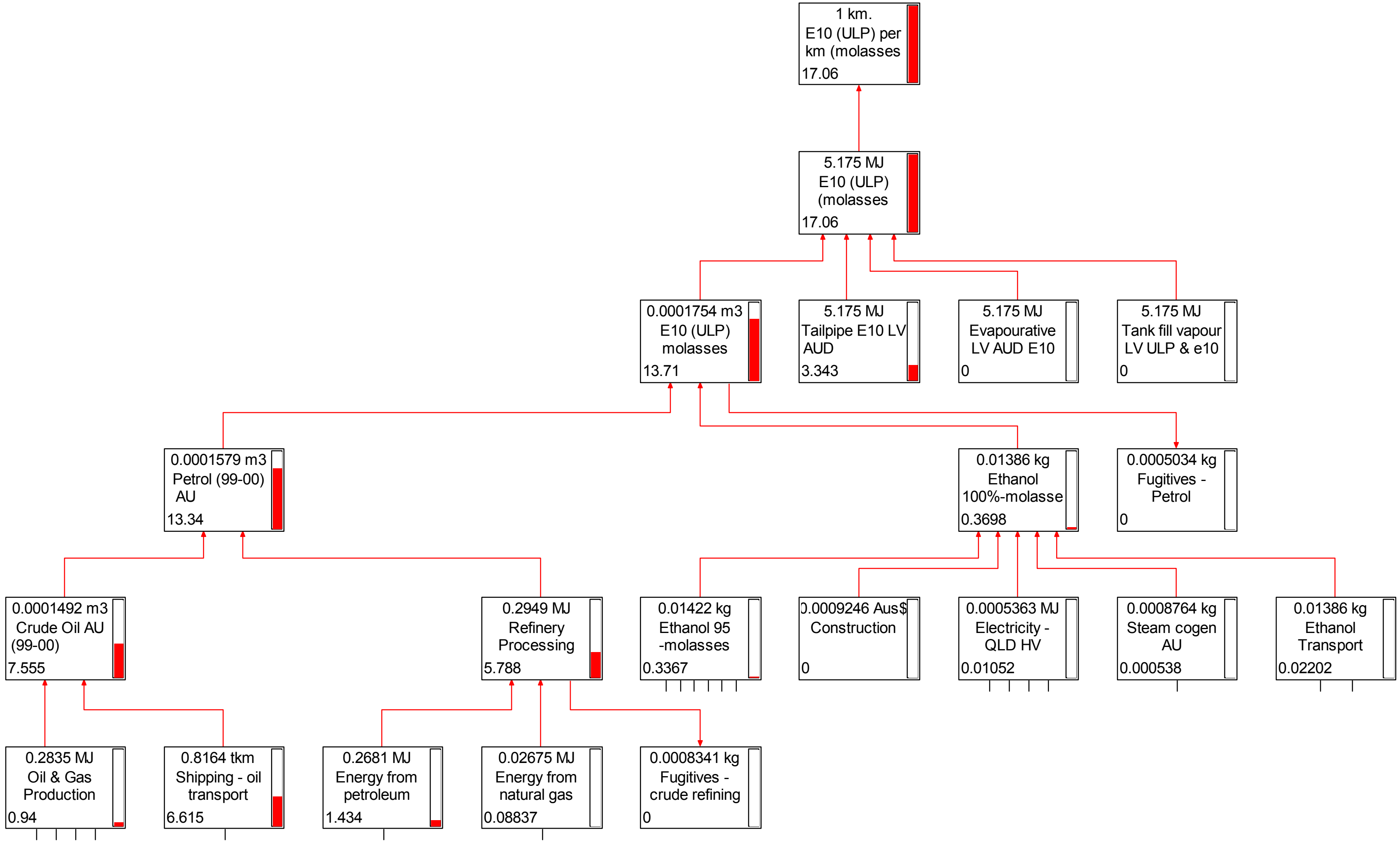


Figure A59. Ethanol (E10 – molasses co-generation) emissions in mg PM per km (standardised passenger car)

Sorghum

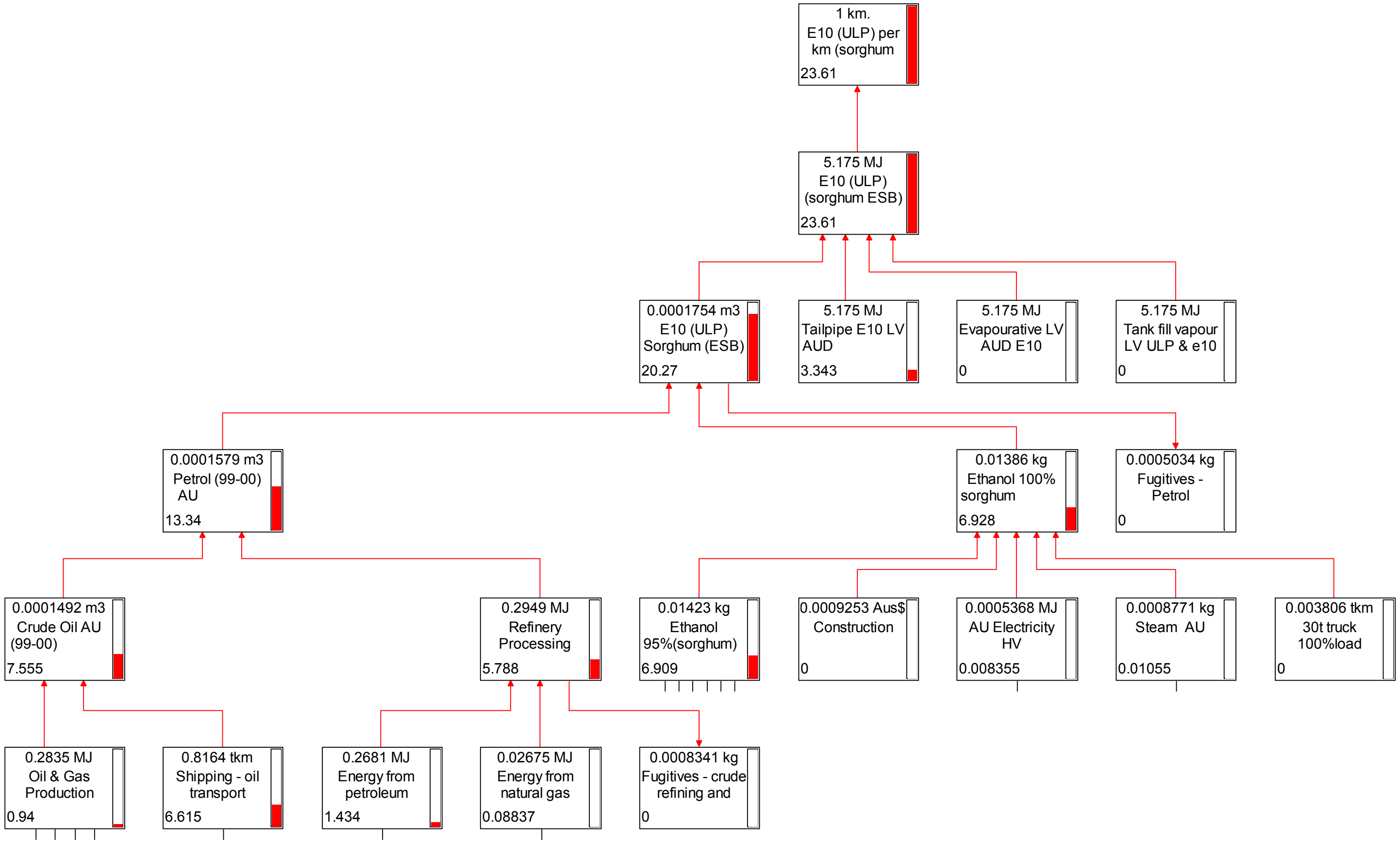


Figure A60. Ethanol (E10 - sorghum) emissions in mg PM per km (standardised passenger car)

Wheat

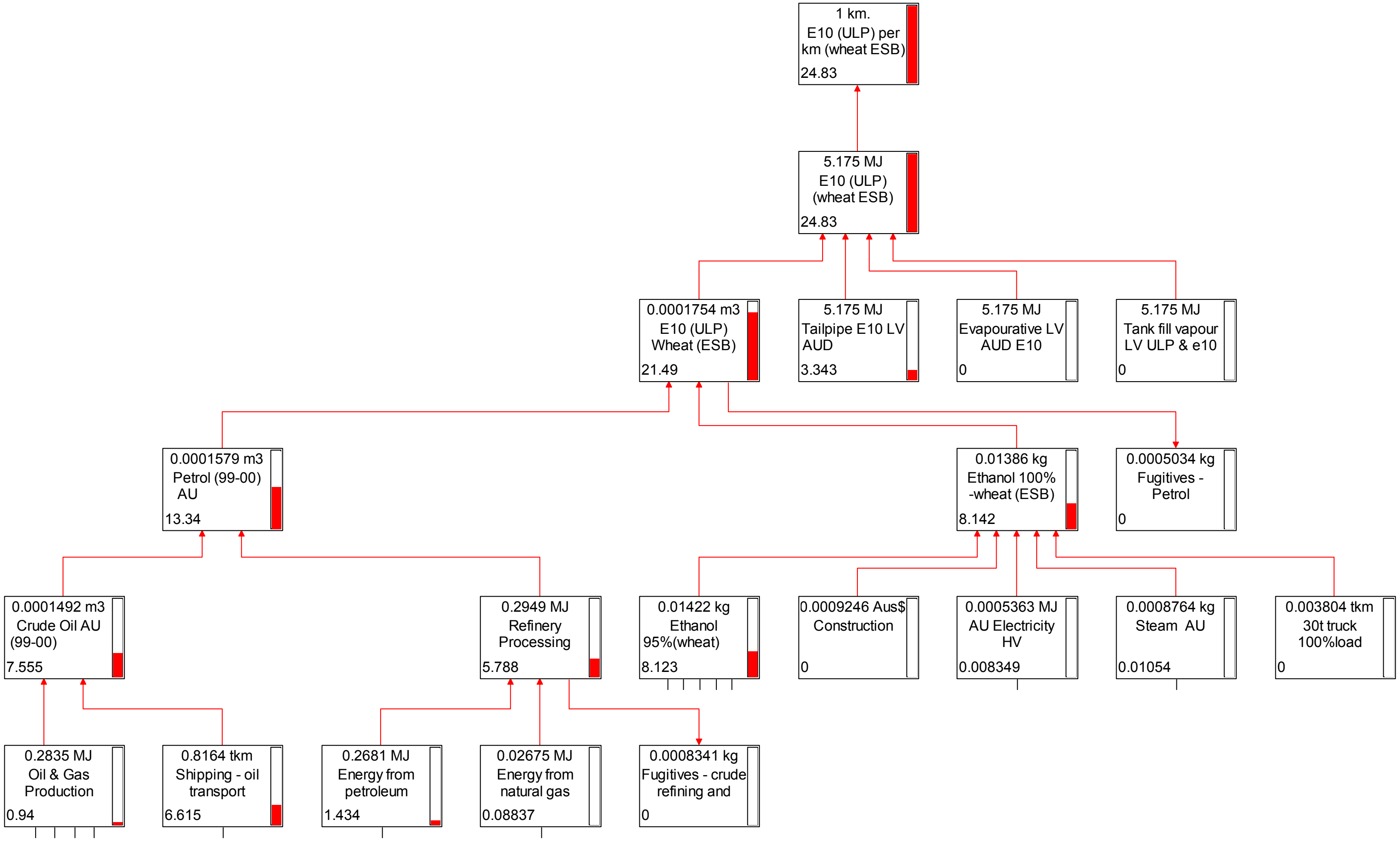


Figure A61. **Ethanol (E10 - wheat) emissions in mg PM per km (standardised passenger car)**

Wheat starch waste

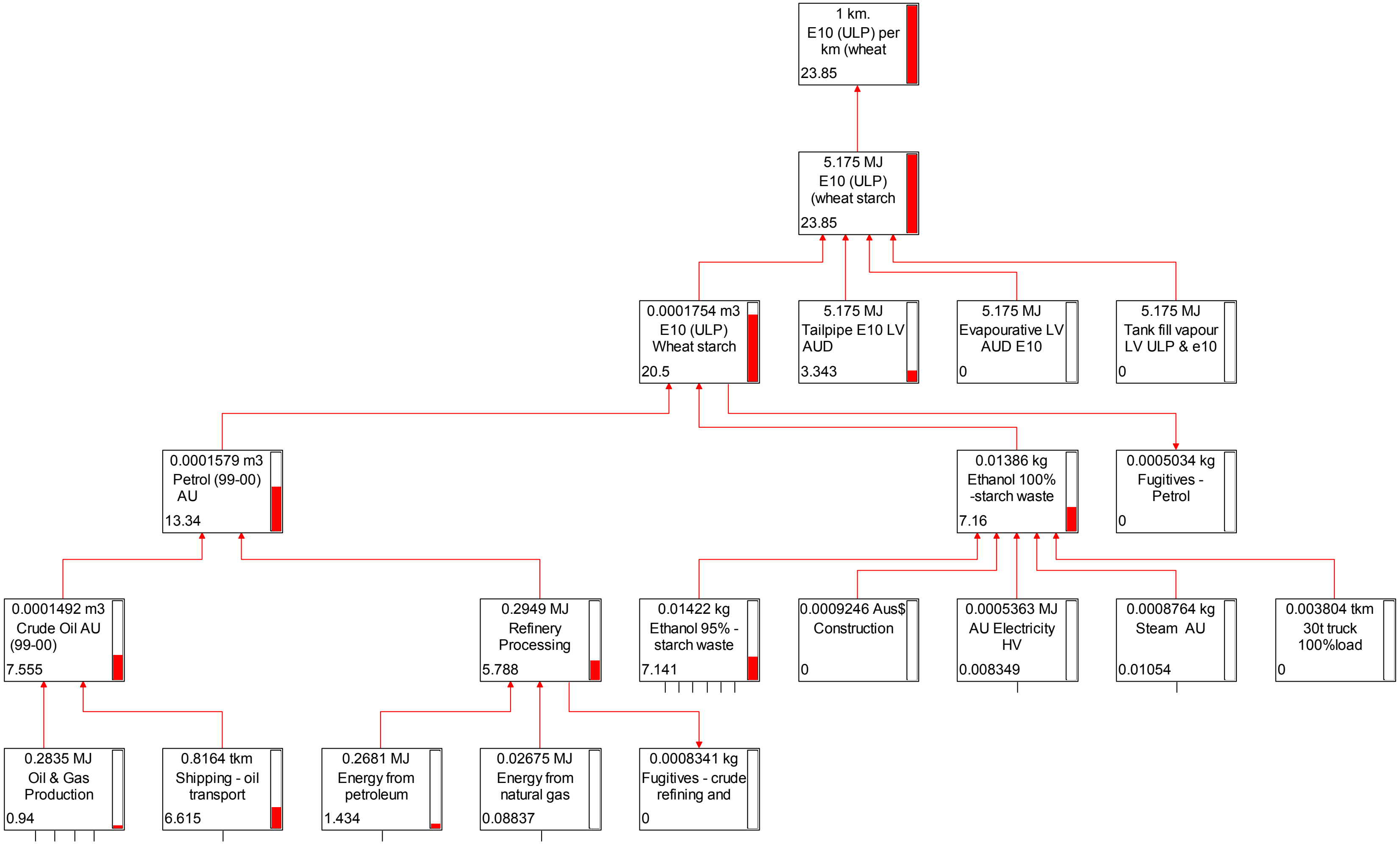


Figure A62. Ethanol (E10 – wheat starch waste) emissions in mg PM per km (standardised passenger car)

Fuel Life Cycle (WTW) GHG Emissions per km - Diesel and 100% Biodiesel

LSD

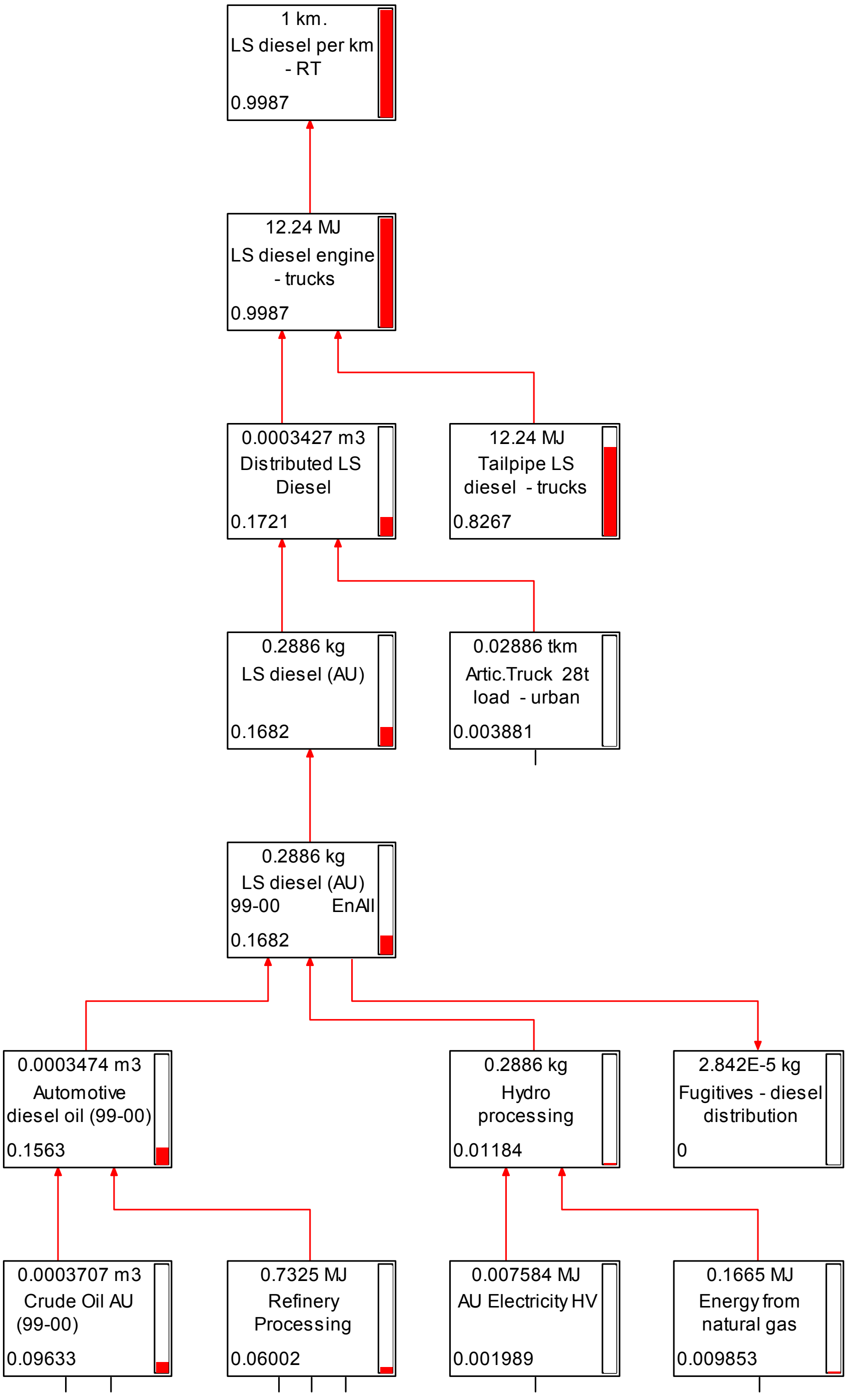


Figure A63. **Low-sulfur diesel (LSD) emissions in kg CO_2-e per km (rigid truck)**

ULS

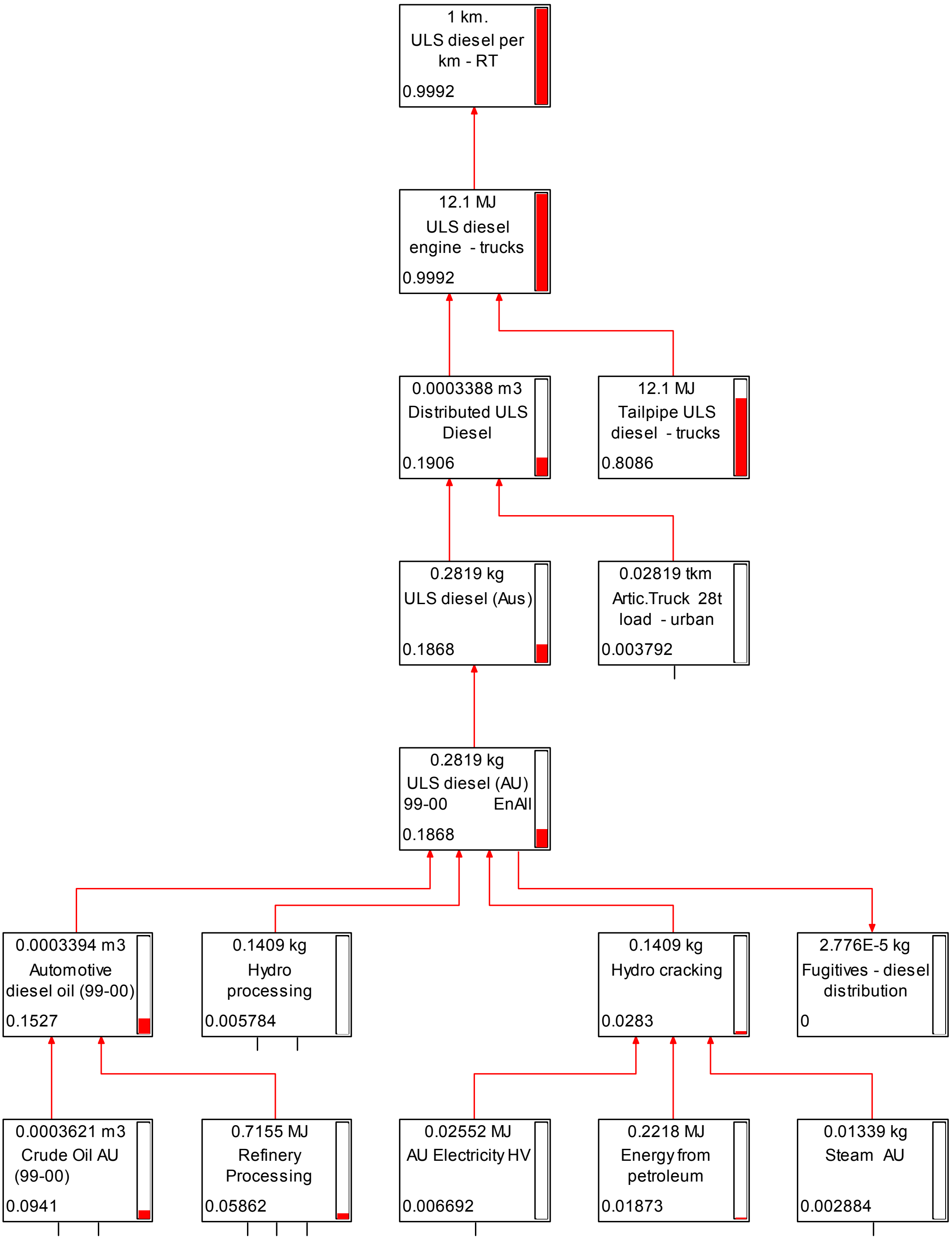


Figure A64. Ultra low-sulfur diesel (ULS) emissions in kg CO_2-e per km (rigid truck)

XLS

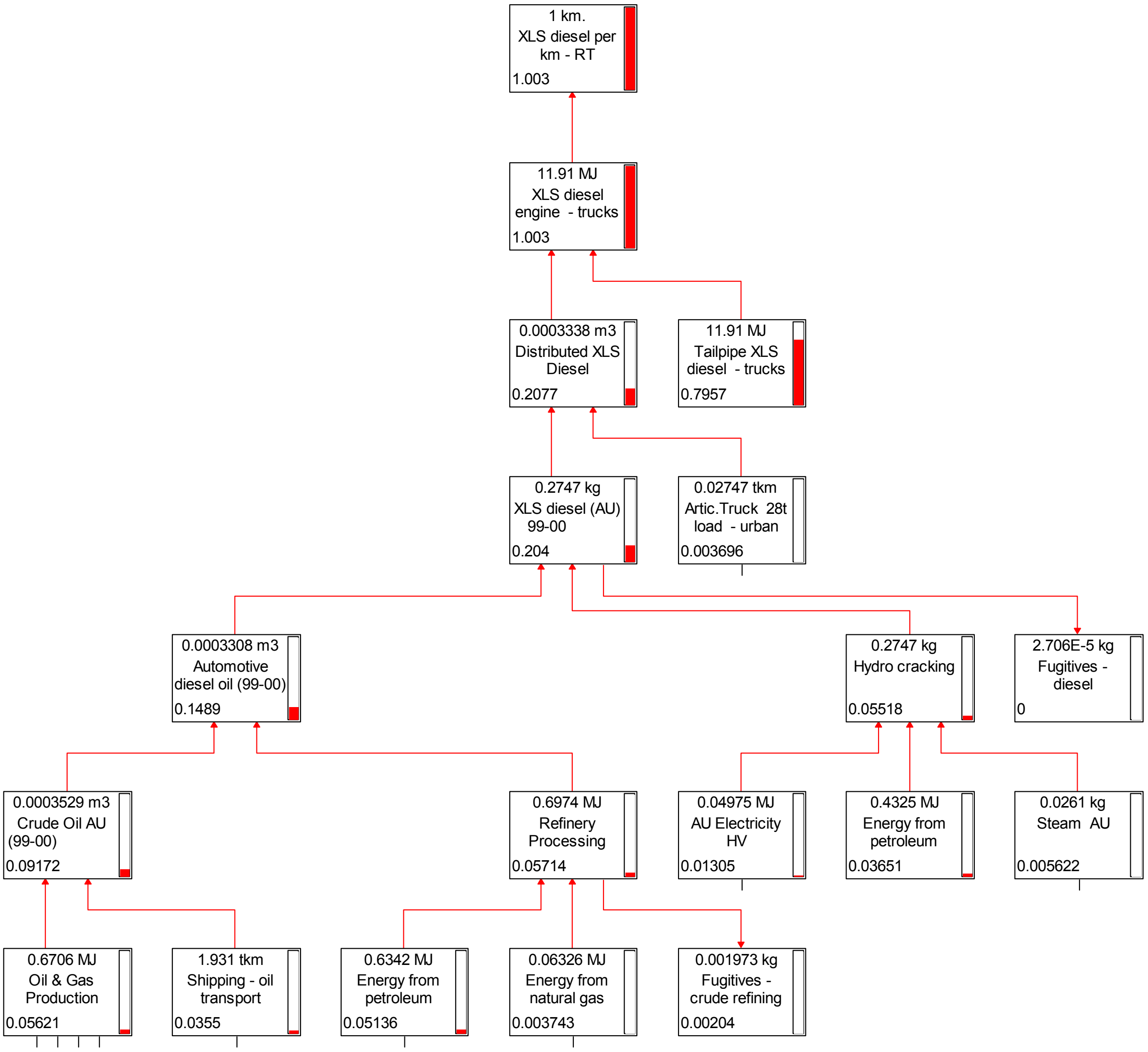


Figure A65. **Extra low-sulfur diesel (XLS) emissions in kg CO_2-e per km (rigid truck)**

Biodiesel canola

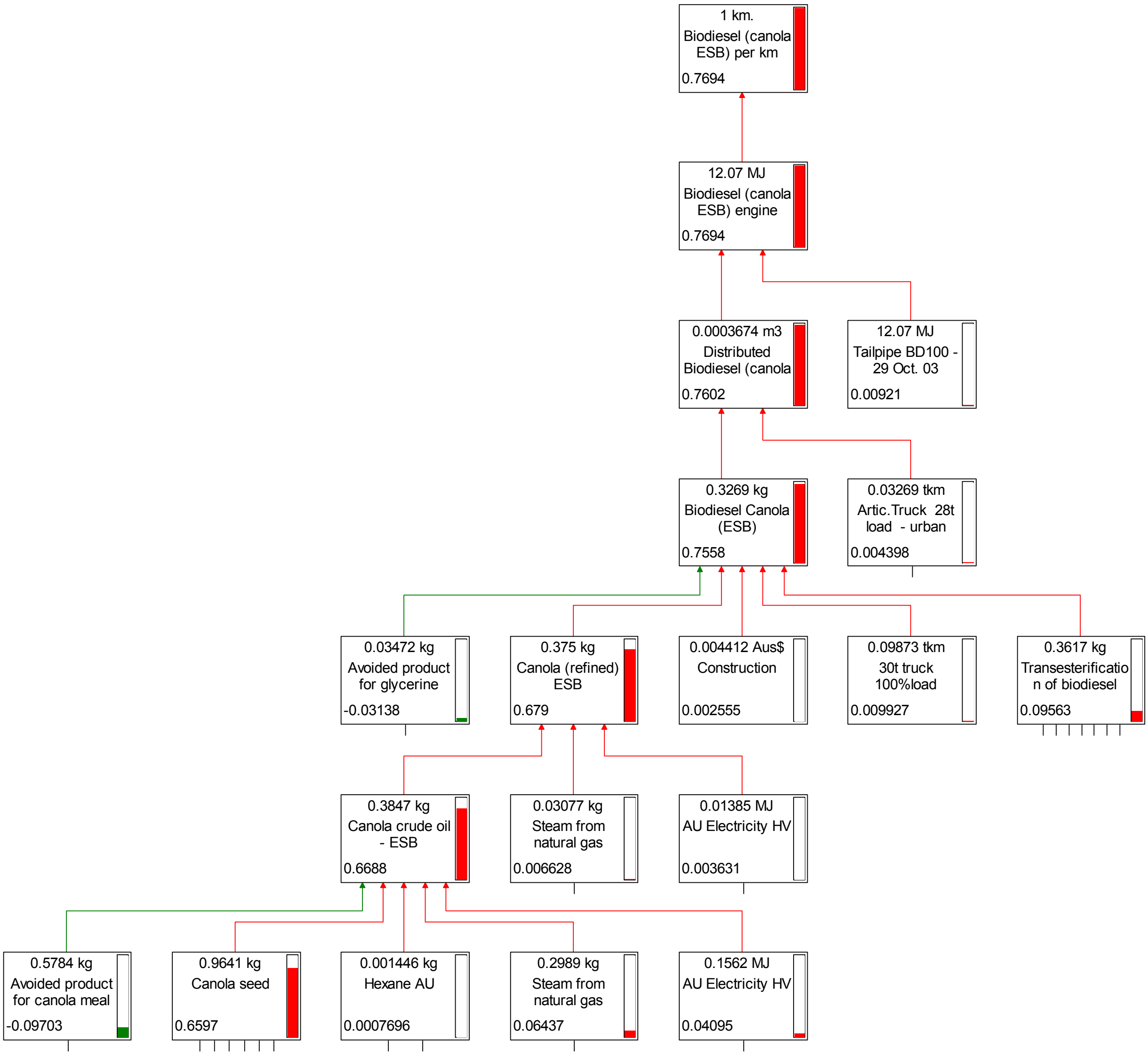


Figure A66. Biodiesel (BD100 - canola) emissions in kg CO_2-e per km (rigid truck)

Biodiesel tallow

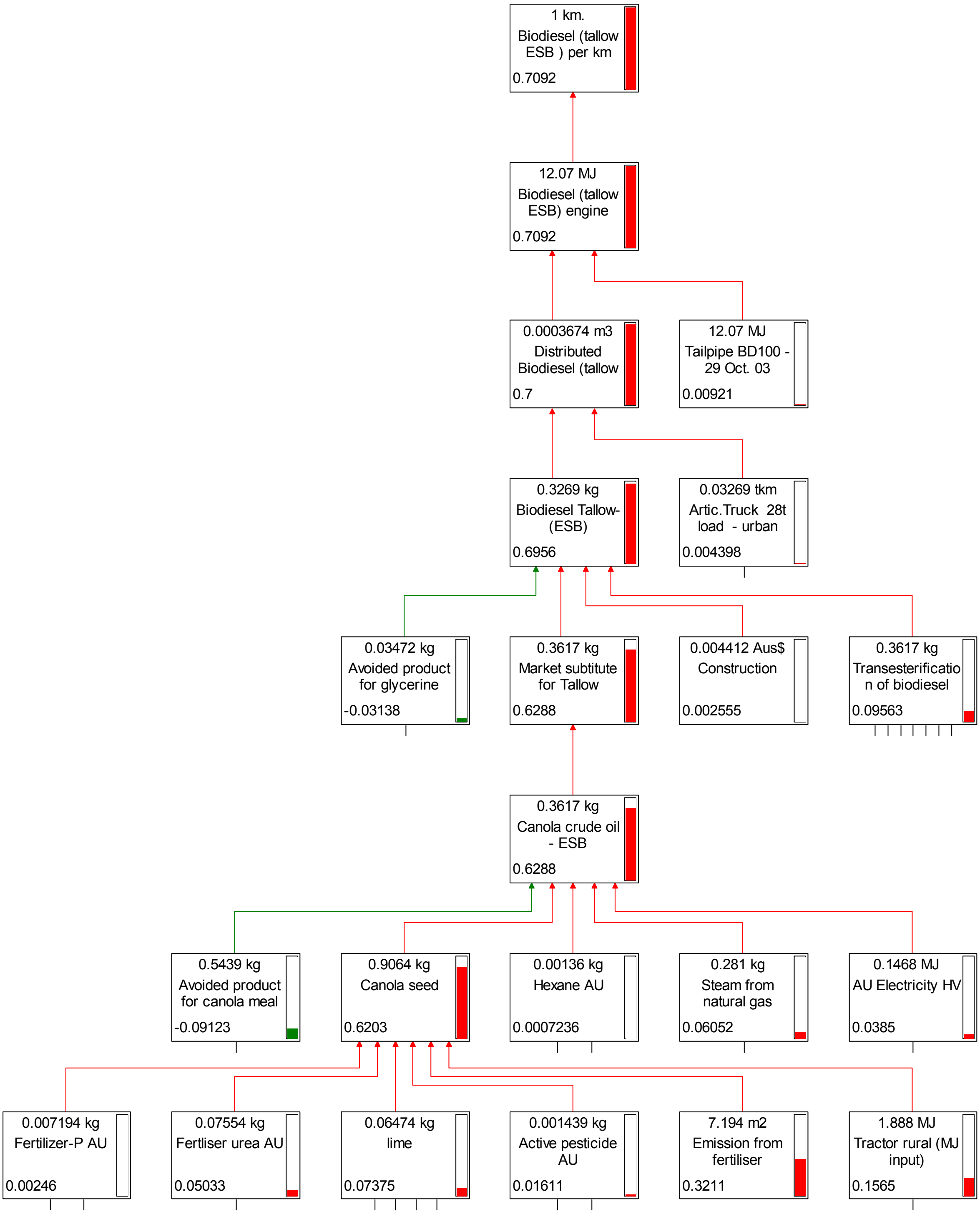


Figure A67. **Biodiesel (BD100 - tallow) emissions in kg CO_2-e per km (rigid truck)**

Biodiesel waste oil

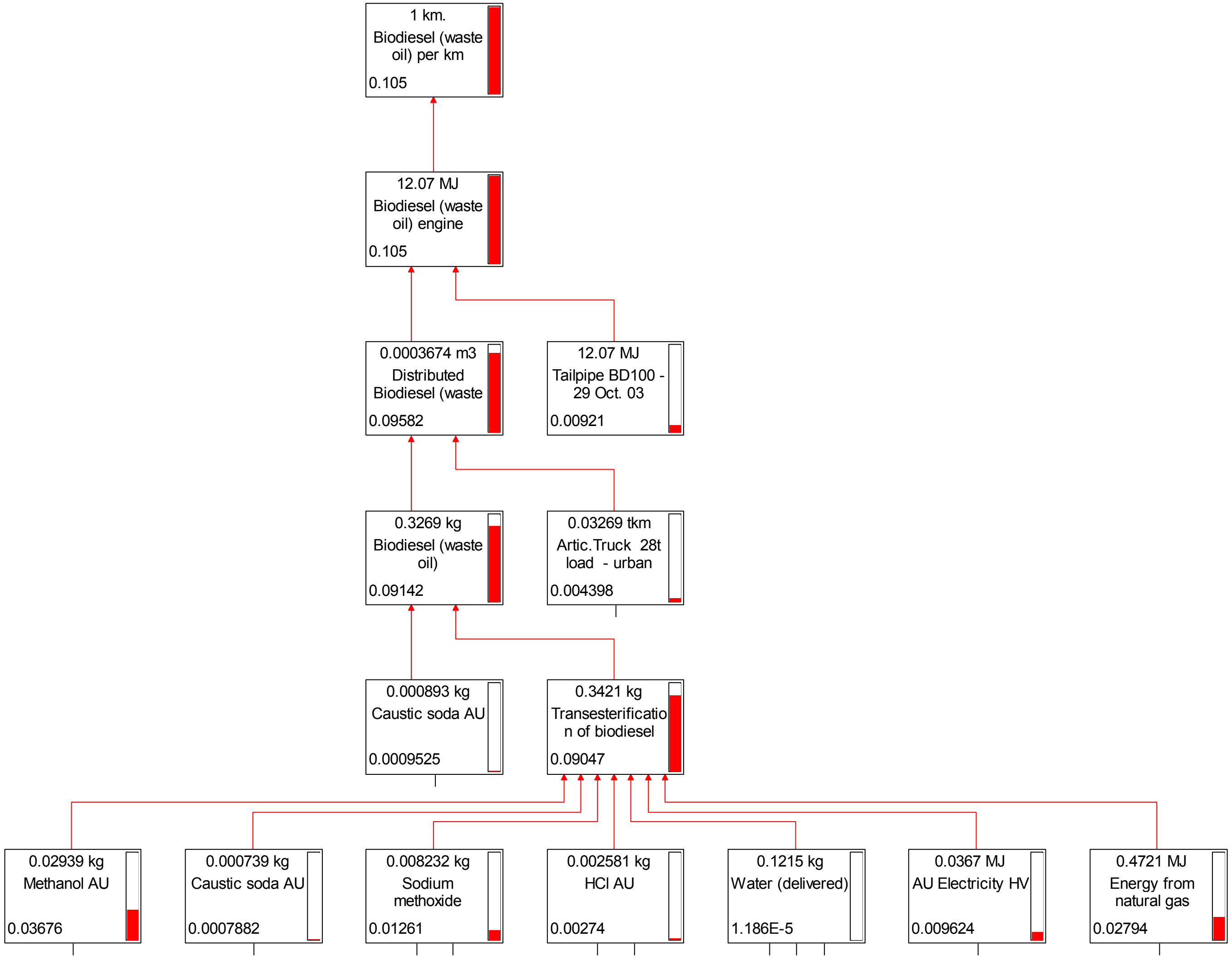


Figure A68. Biodiesel (BD100 - waste oil) emissions in kg CO_2-e per km (rigid truck)

Full Life Cycle (WTW) Air Pollutant Emissions per km – Diesel and 100% Biodiesel

Upstream NO_x emissions per km

LSD

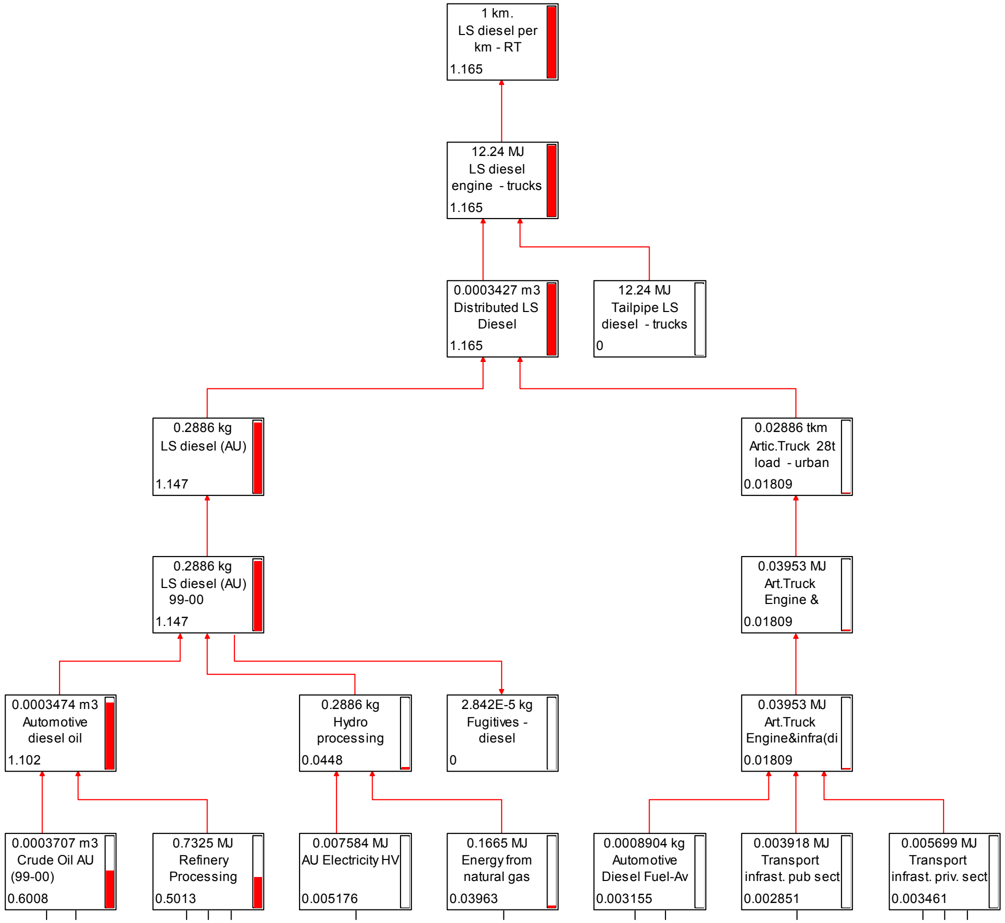


Figure A69. Low sulfur diesel (LSD) emissions in g NO_x per km (rigid truck)

ULS

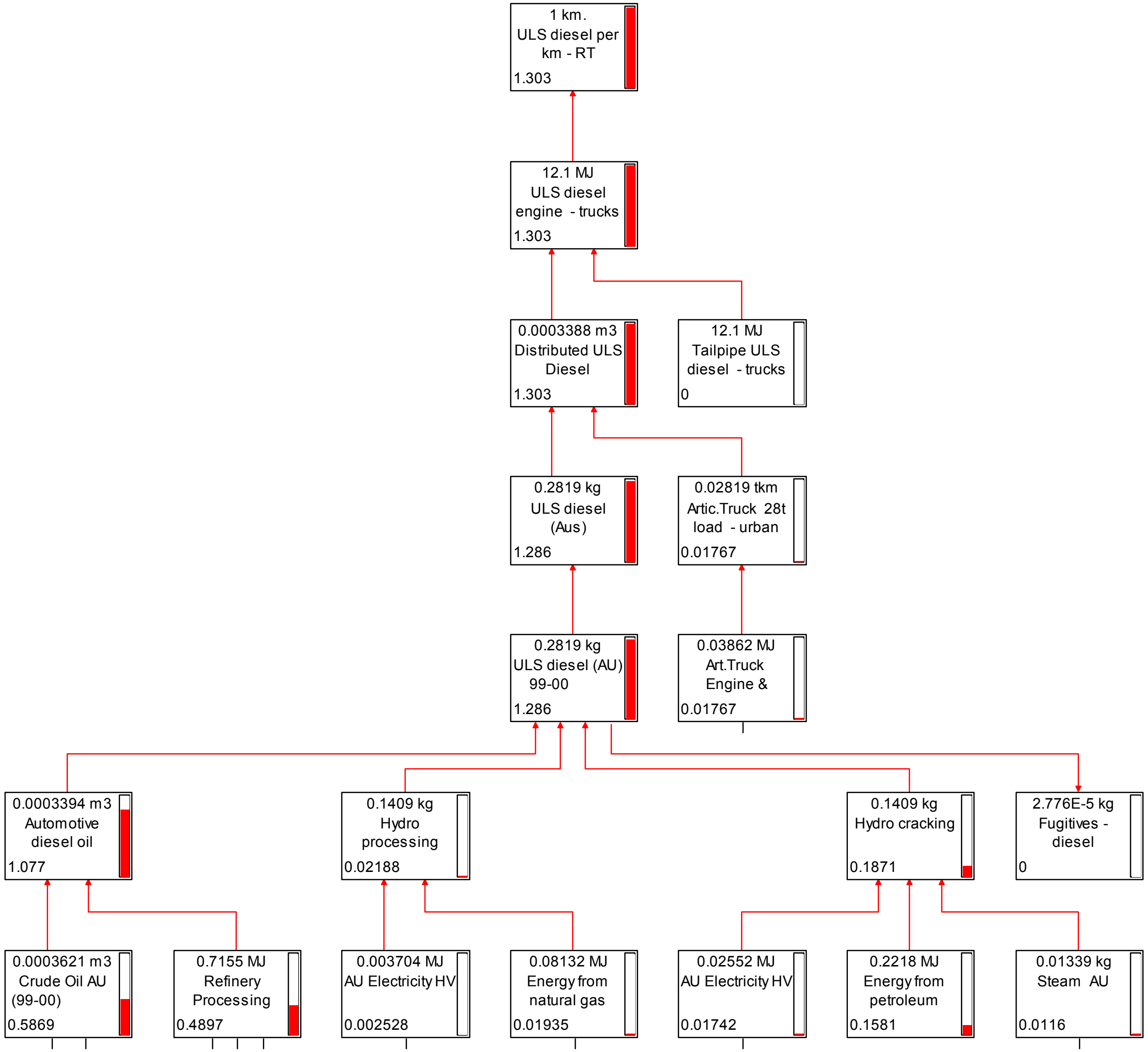


Figure A70. Ultra-low sulfur diesel (ULS) emissions in g NO_x per km (rigid truck)

XLS

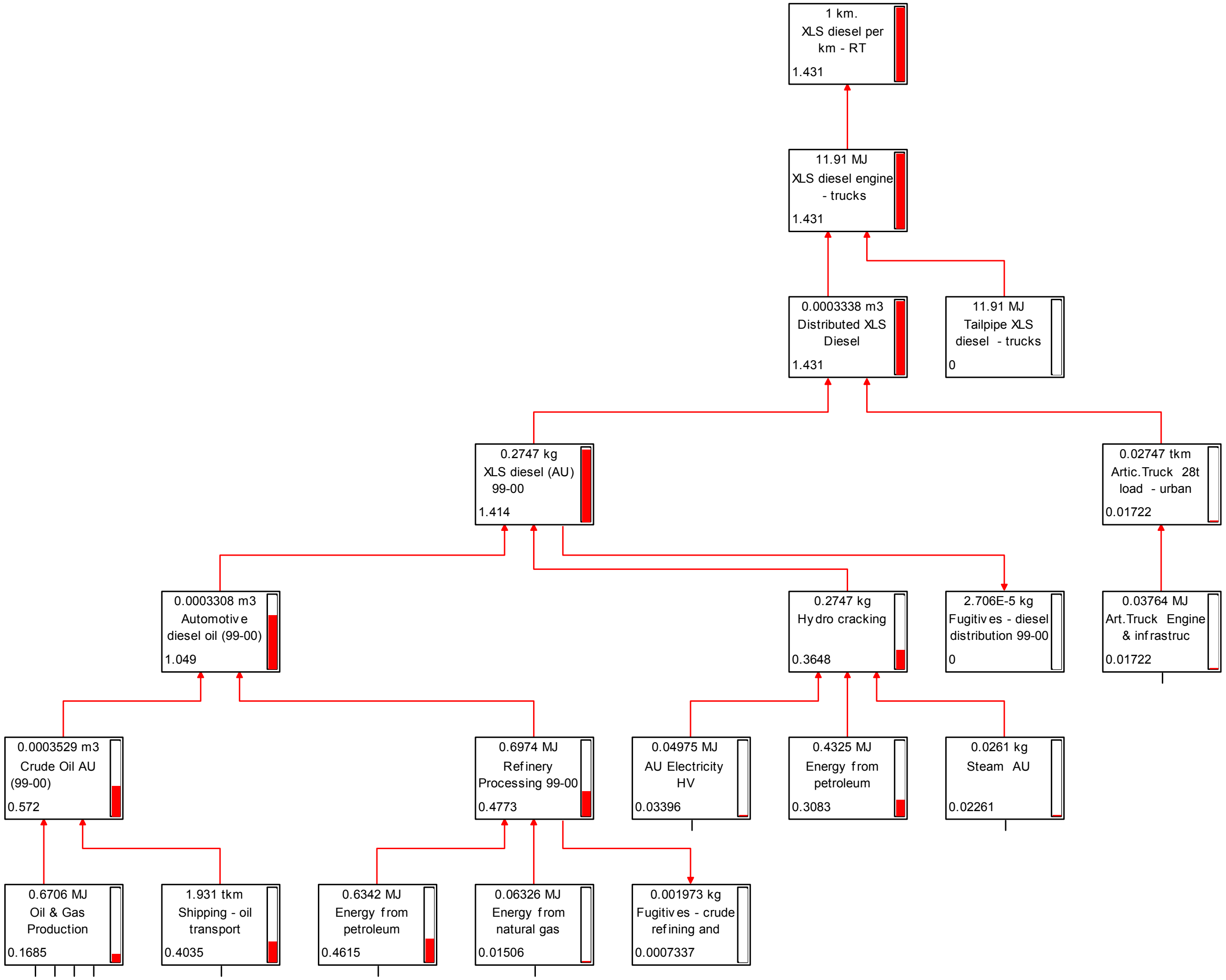


Figure A71. Extra-low sulfur diesel (XLS) emissions in g NO_x per km (rigid truck)

Biodiesel canola

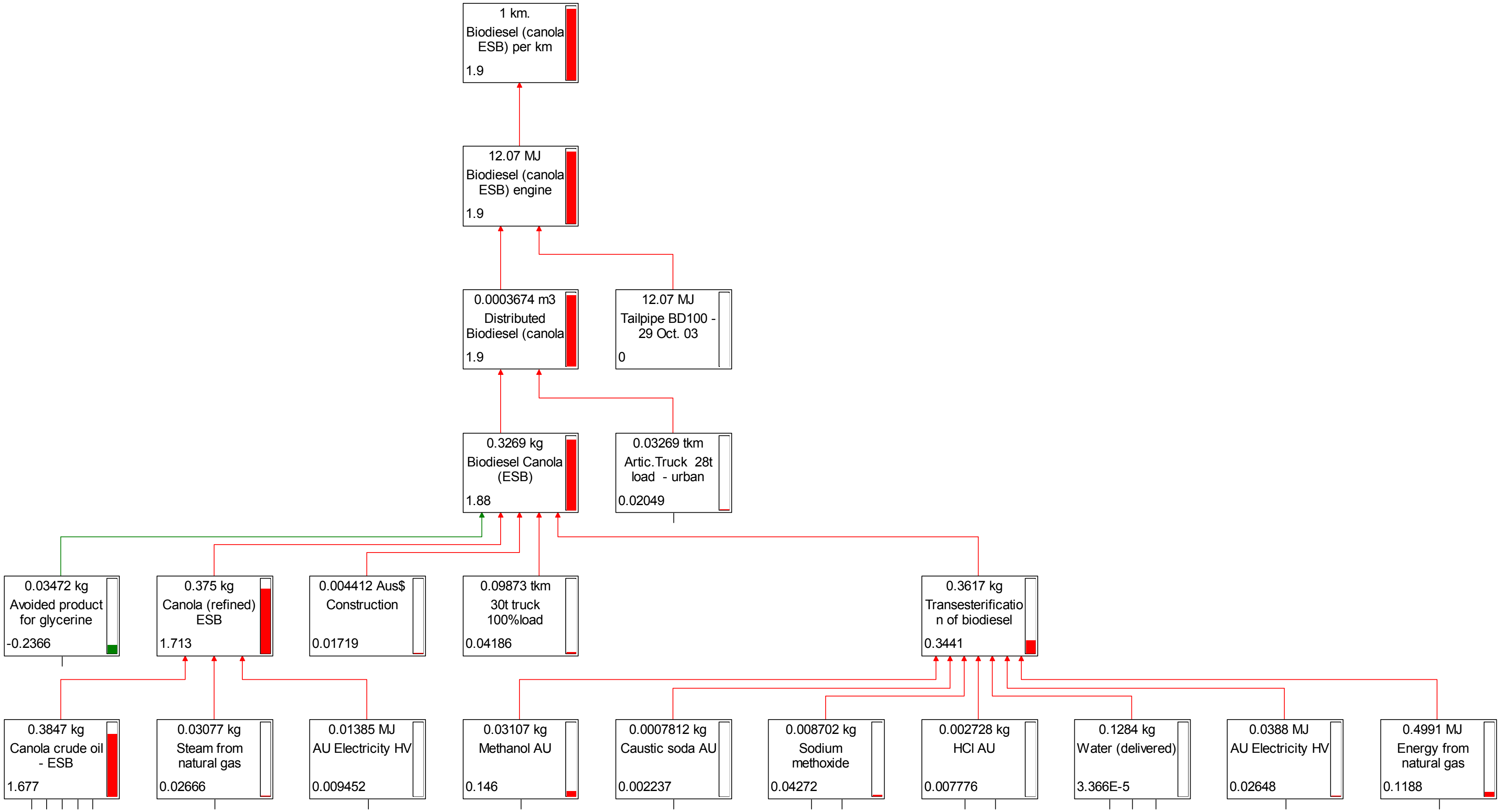


Figure A72. **Biodiesel (BD100 - canola) emissions in g NO_x per km (rigid truck)**

Biodiesel tallow

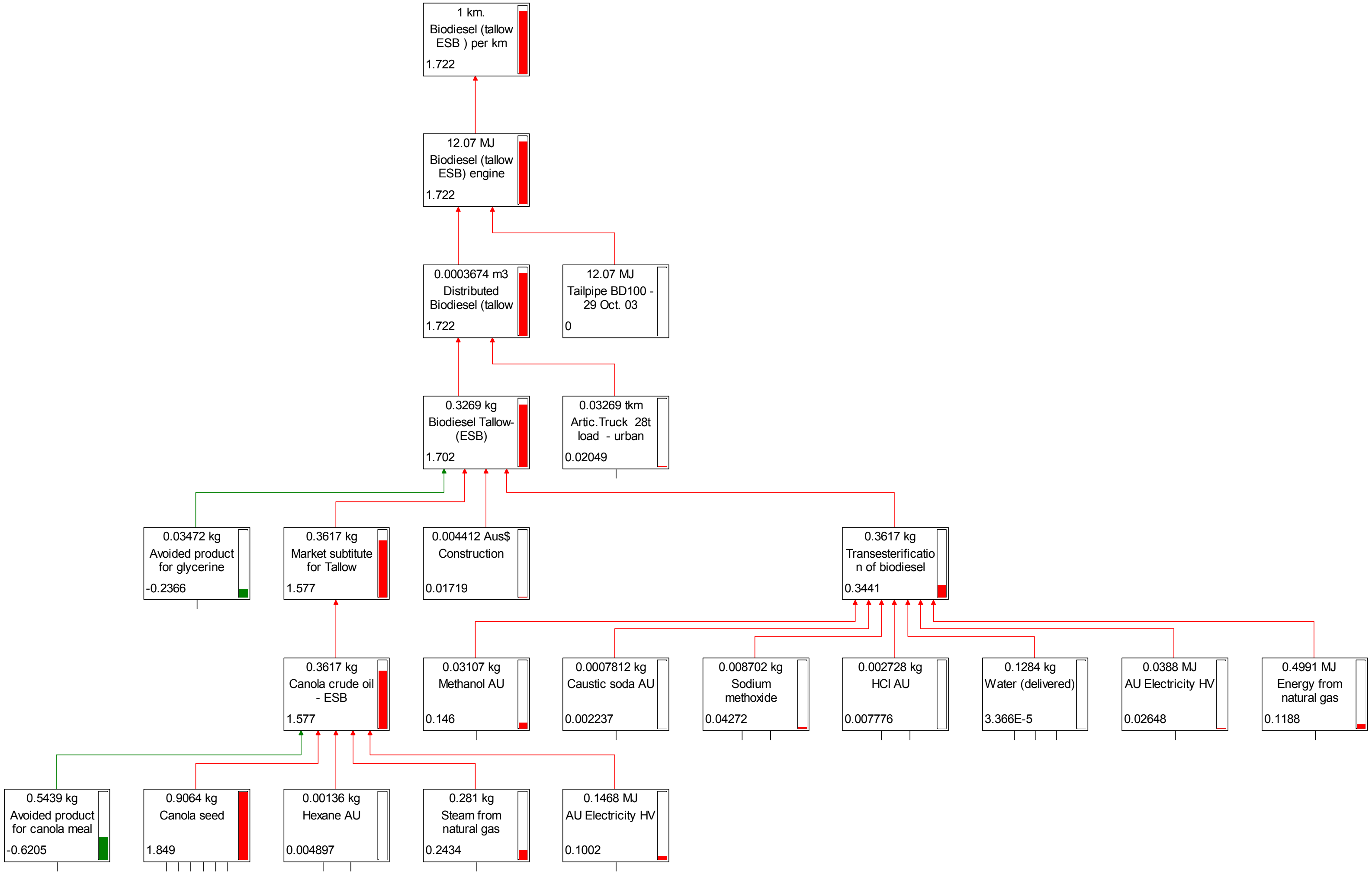


Figure A73. **Biodiesel (BD100 - tallow) emissions in g NO_x per km (rigid truck)**

Biodiesel waste oil

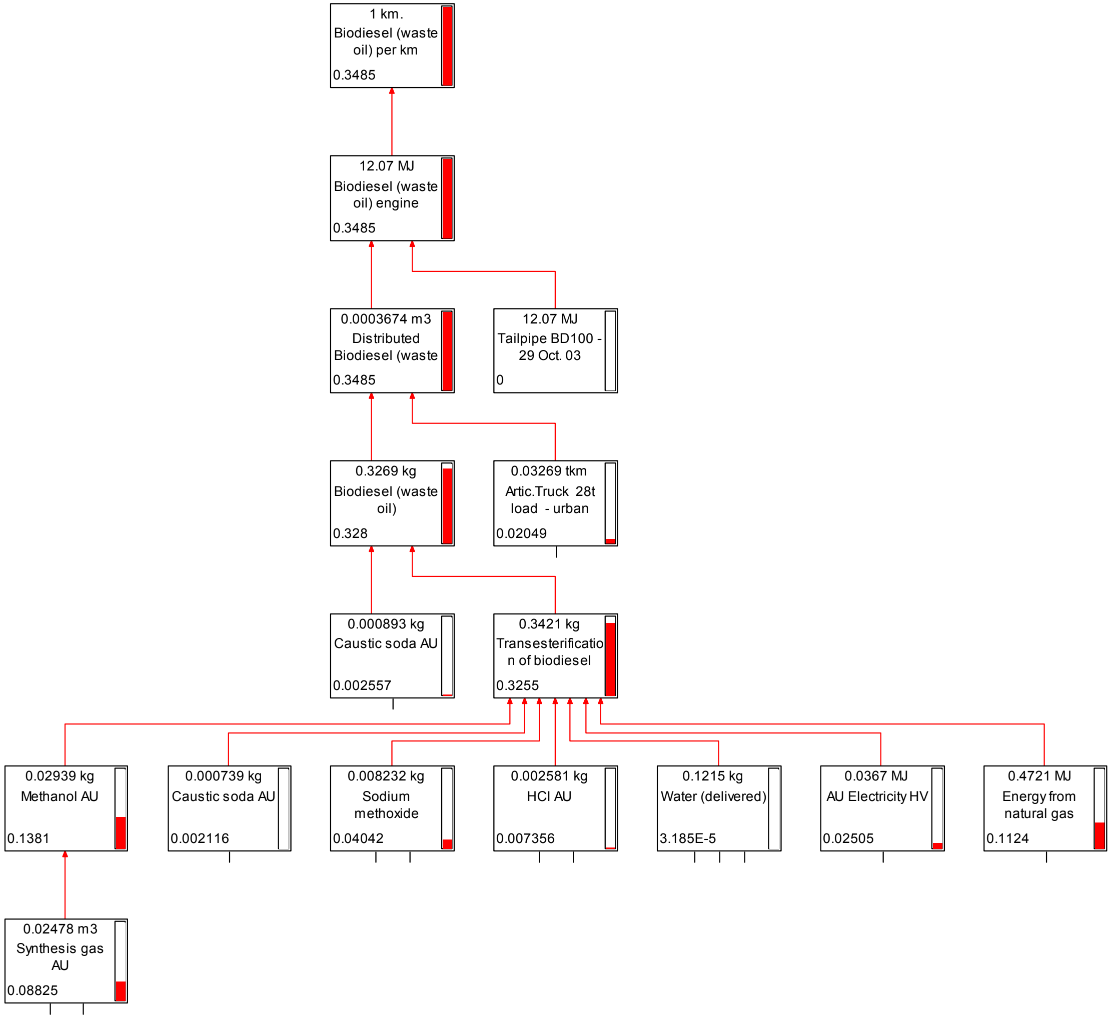


Figure A74. **Biodiesel (BD100 – waste oil) emissions in g NO_x per km (rigid truck)**

Upstream NMHC Emissions per km

LSD

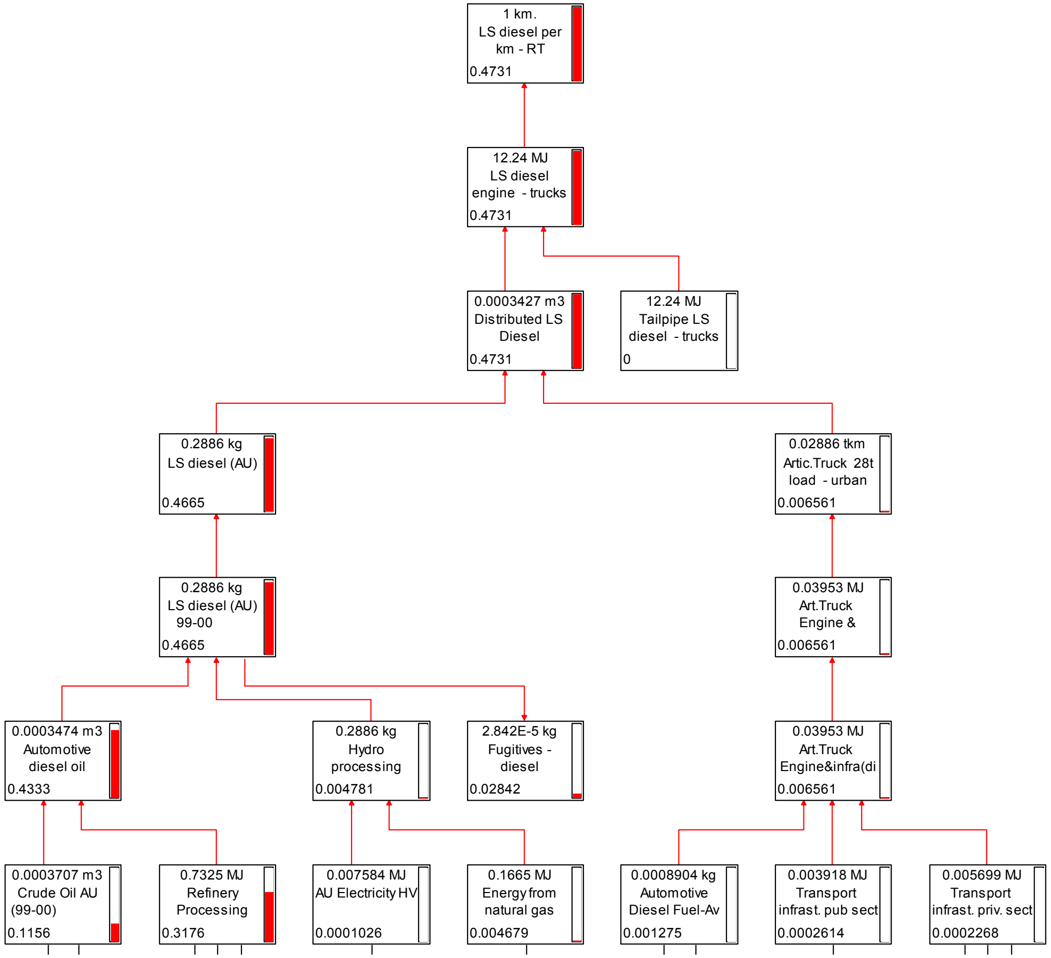


Figure A75. Low-sulfur diesel (LSD) emissions in g NMHC per km (rigid truck)

ULS

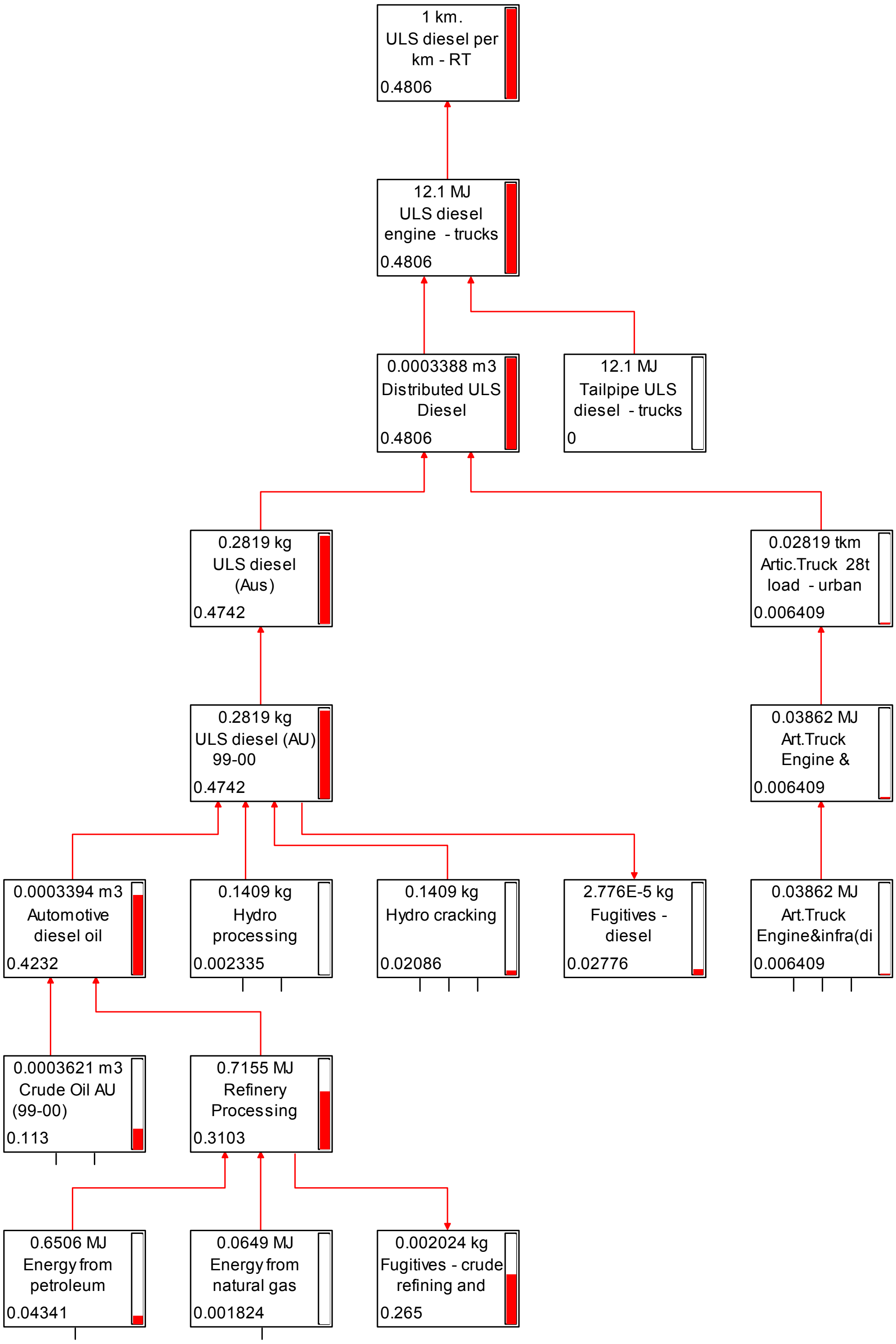


Figure A76. Ultra low-sulfur diesel (ULS) emissions in g NMHC per km (rigid truck)

XLS

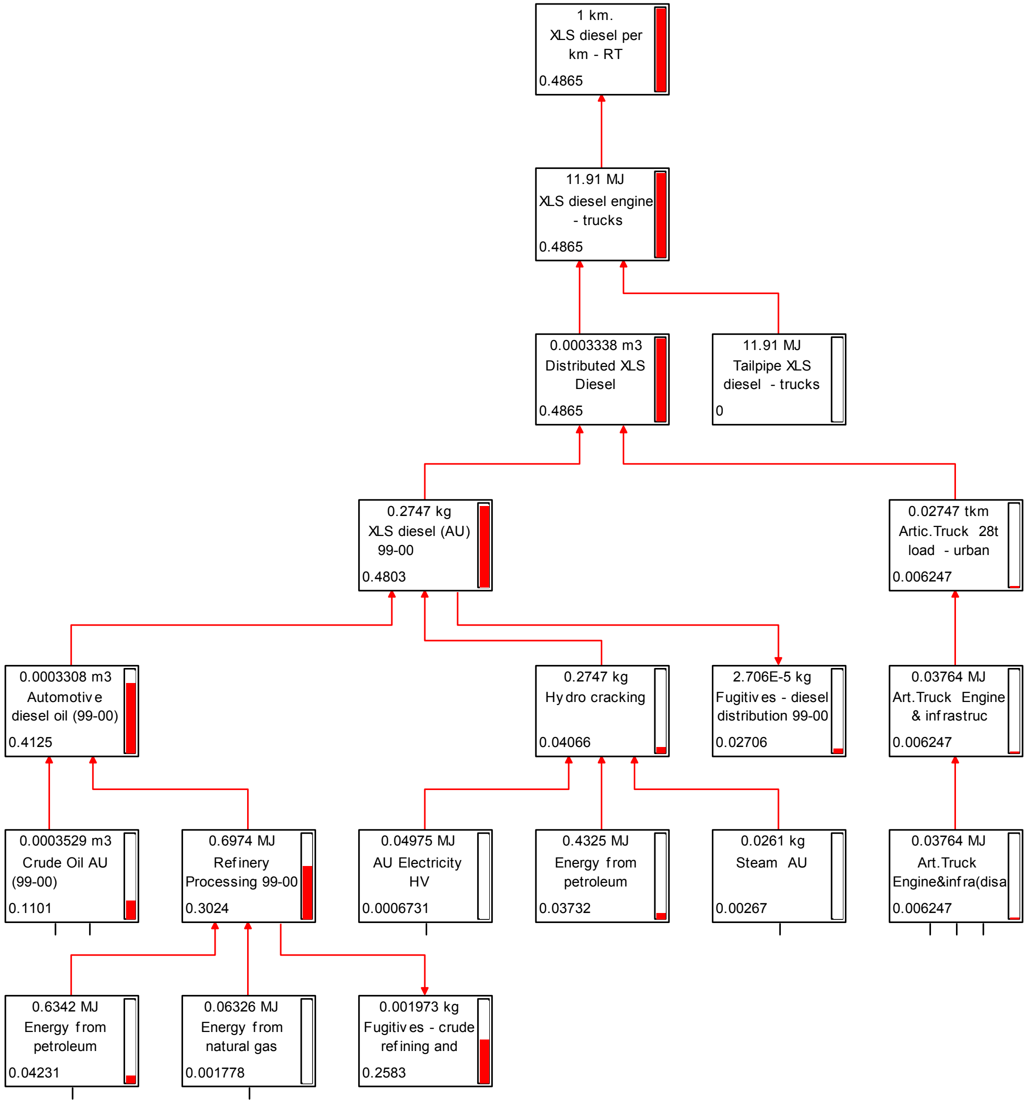


Figure A77. Extra low-sulfur diesel (XLS) emissions in g NMHC per km (rigid truck)

Biodiesel canola

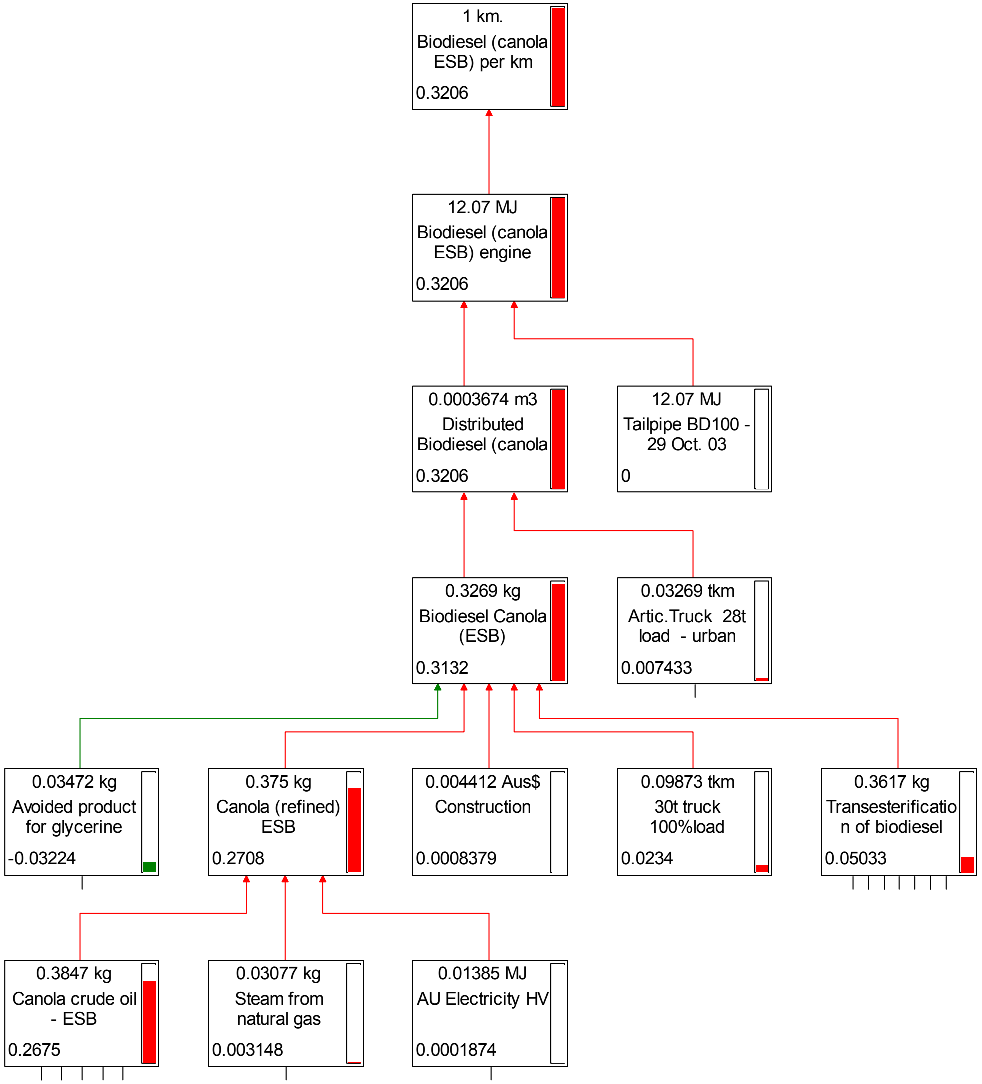


Figure A78. Biodiesel (BD100 - canola) emissions in g NMHC per km (rigid truck)

Biodiesel tallow

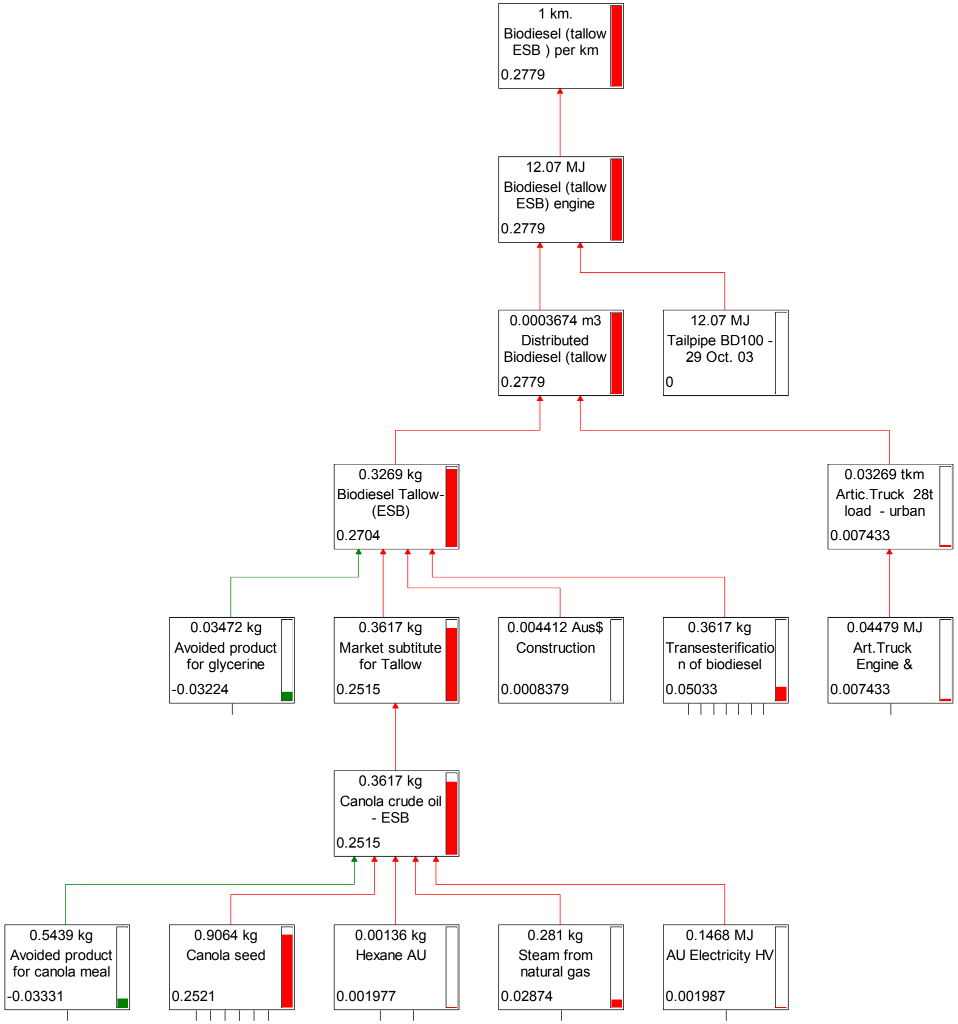


Figure A79. **Biodiesel (BD100 - tallow) emissions in g NMHC per km (rigid truck)**

Biodiesel waste oil

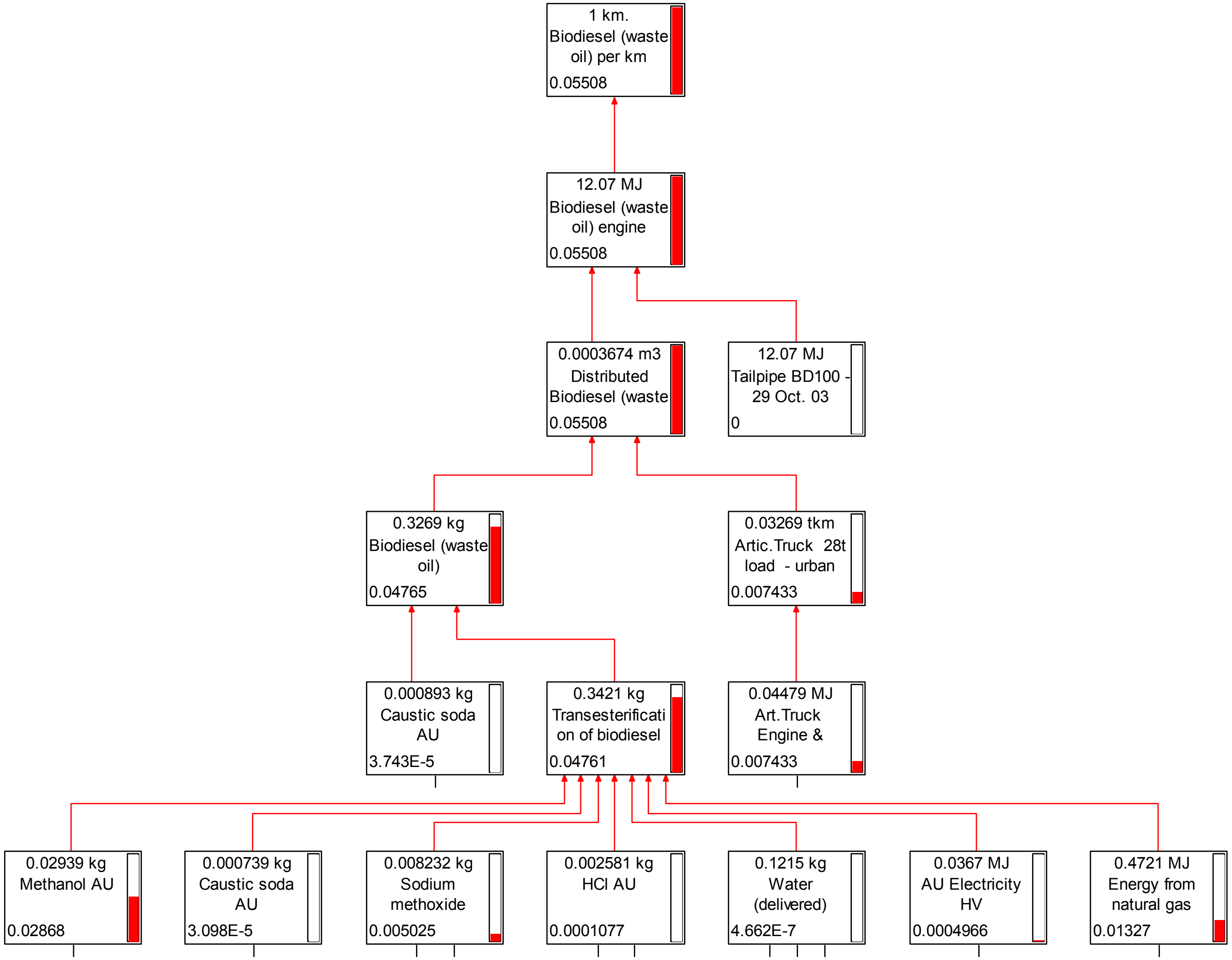


Figure A80. Biodiesel (BD100 – waste oil) emissions in g NMHC per km (rigid truck)

Fuel life cycle PM emissions per km

LSD

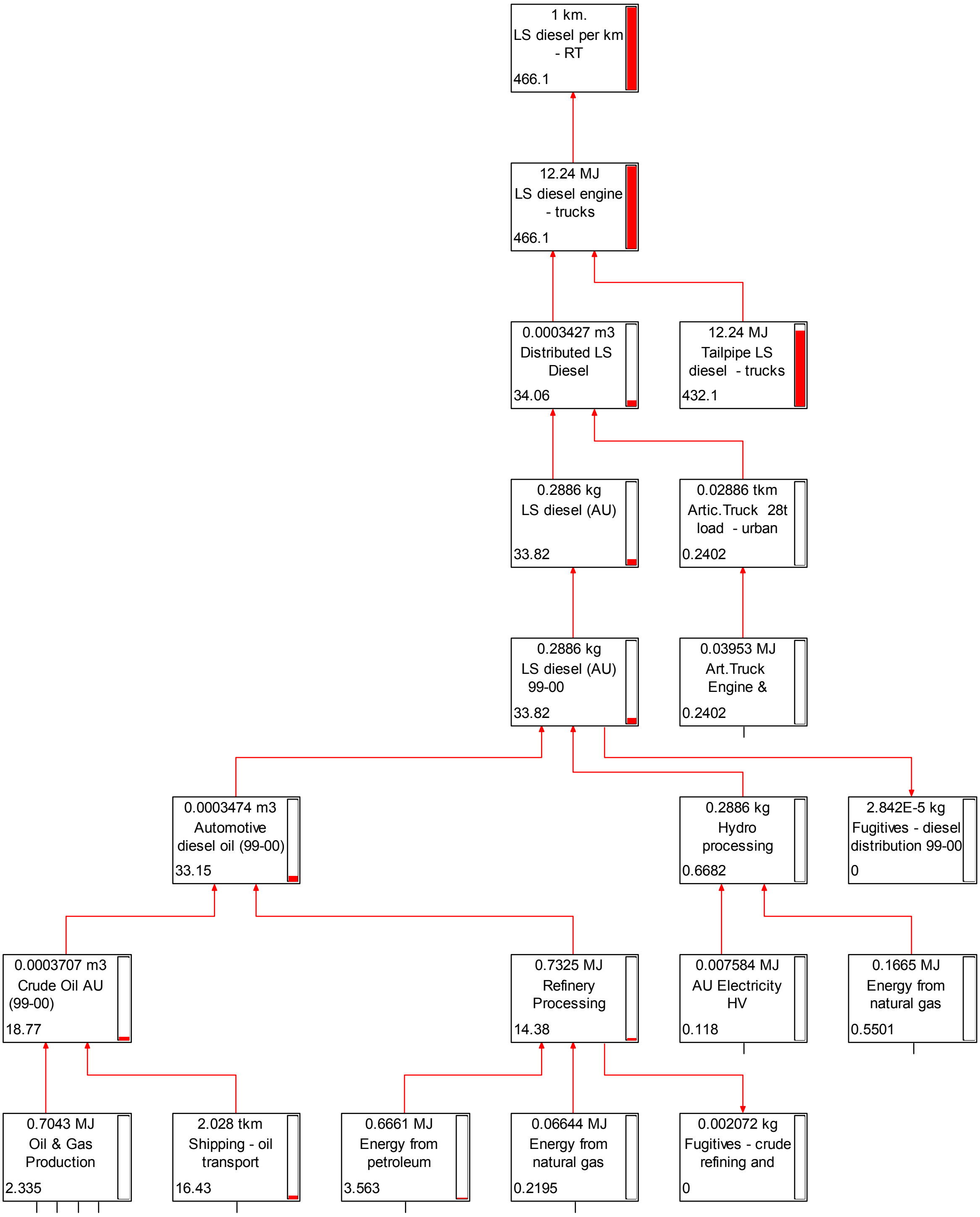


Figure A81. Low-sulfur diesel (LSD) emissions in mg PM per km (rigid truck)

ULS

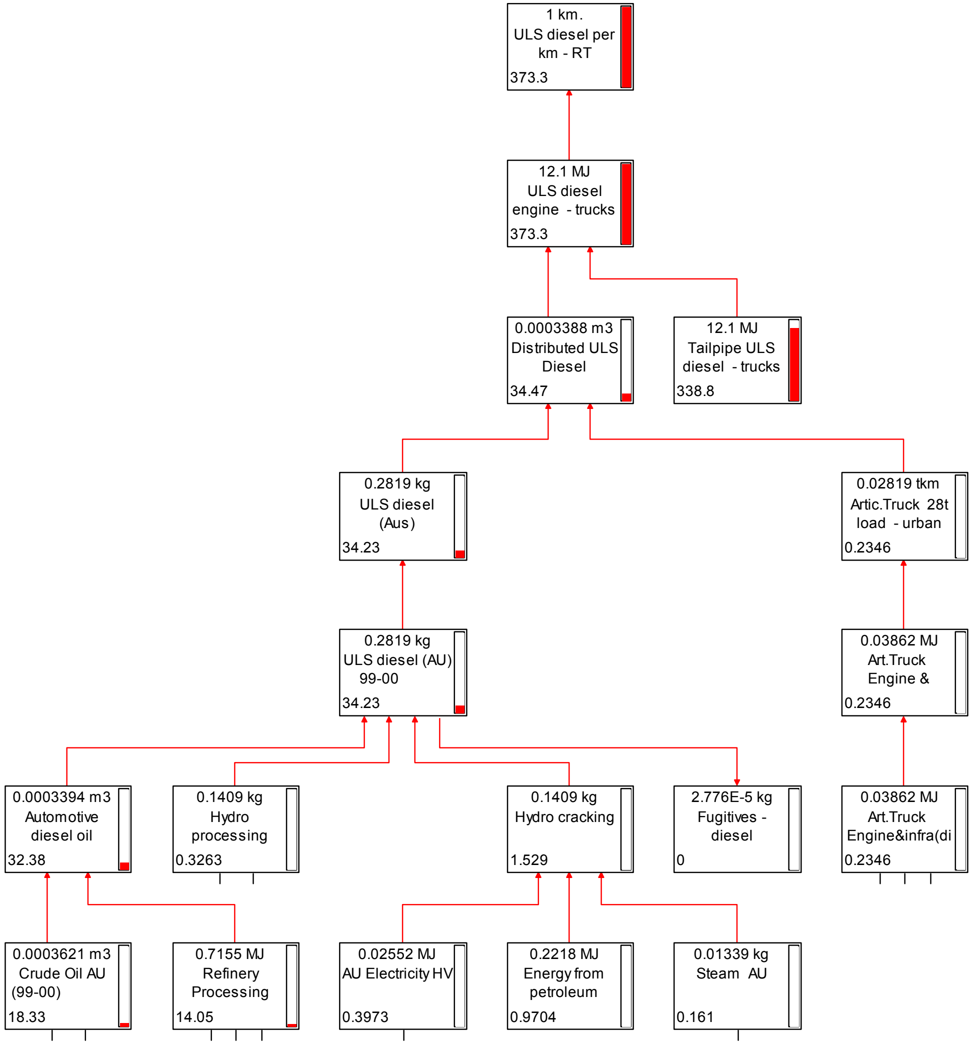


Figure A82. Ultra low-sulfur diesel (ULS) emissions in mg PM per km (rigid truck)

XLS

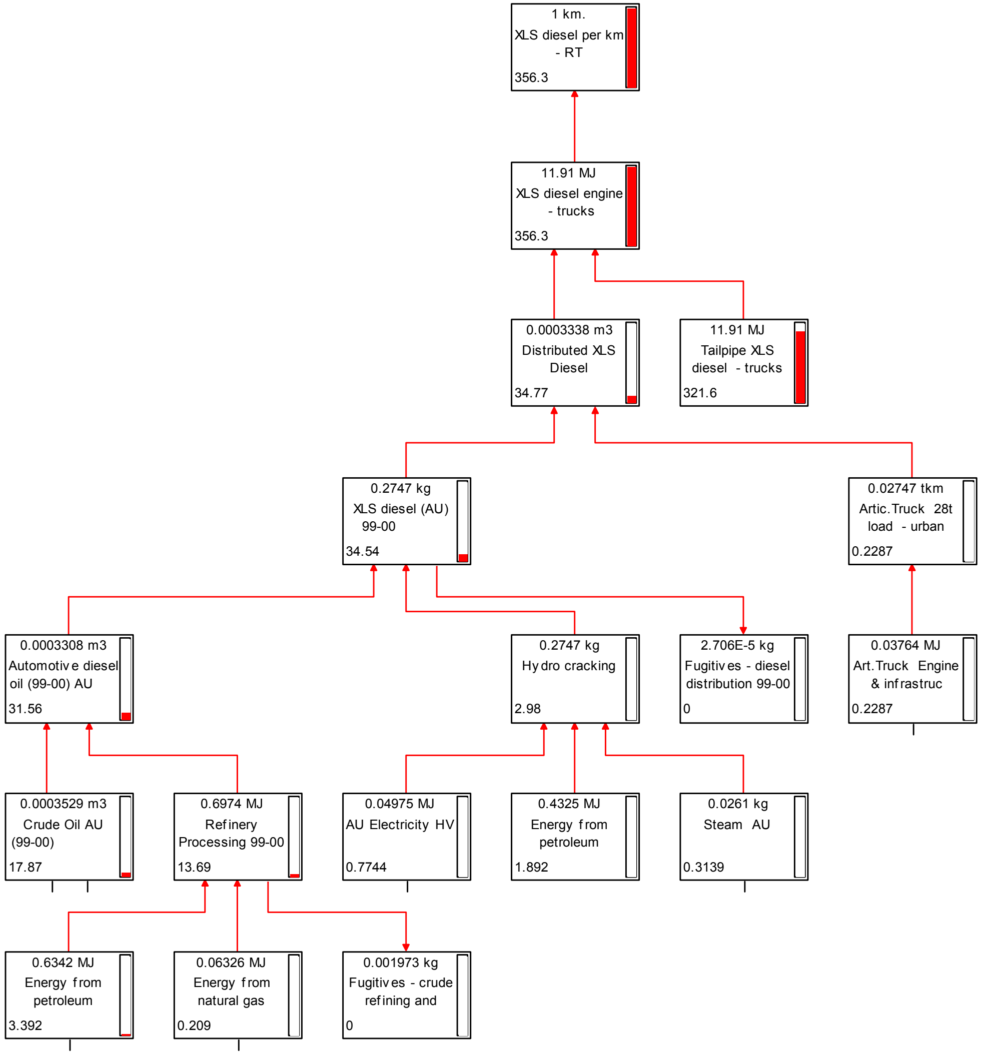


Figure A83. Extra low-sulfur diesel (XLS) emissions in mg PM per km (rigid truck)

Biodiesel canola

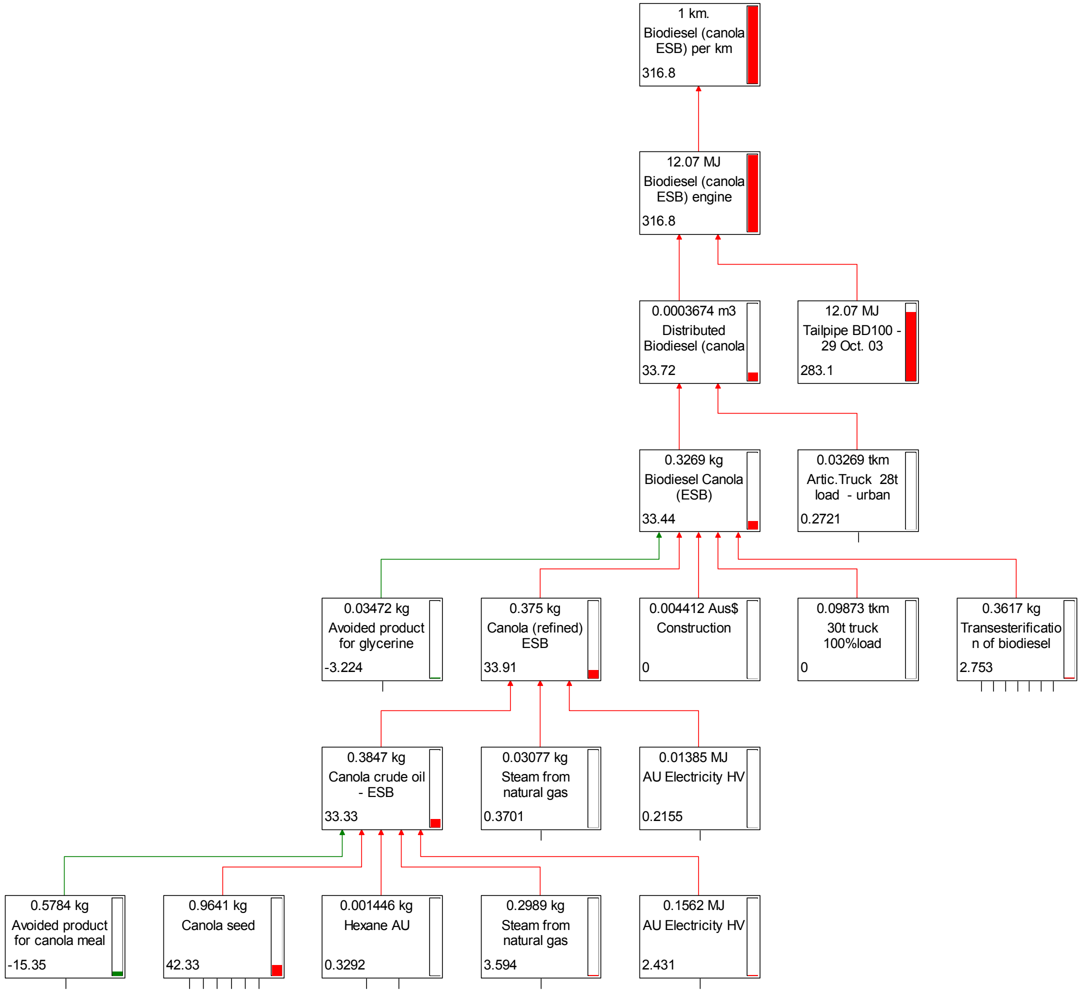


Figure A84. Biodiesel (BD100 - canola) emissions in mg PM per km (rigid truck)

Biodiesel tallow

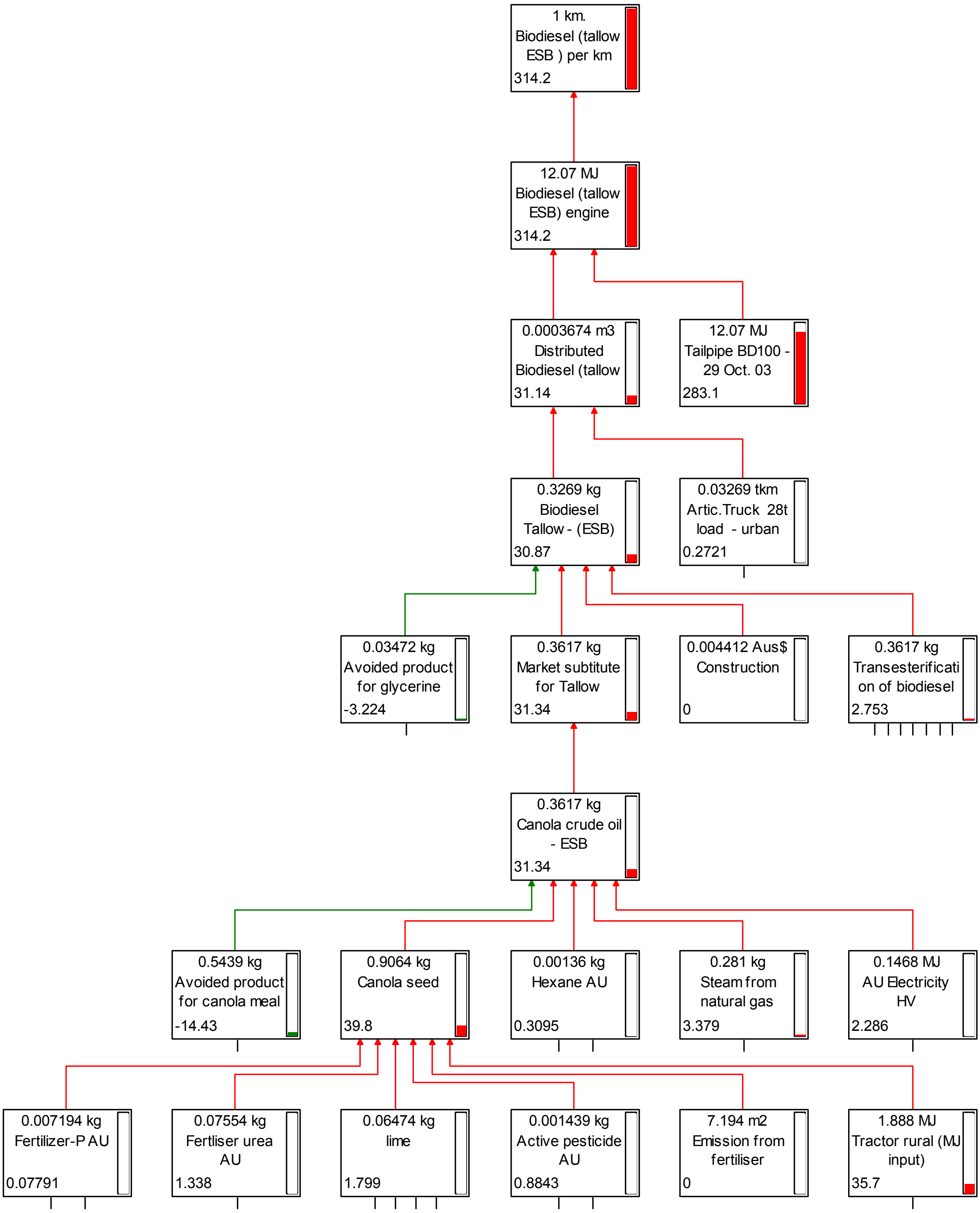


Figure A85. **Biodiesel (BD100 - tallow) emissions in mg PM per km (rigid truck)**

Biodiesel waste oil

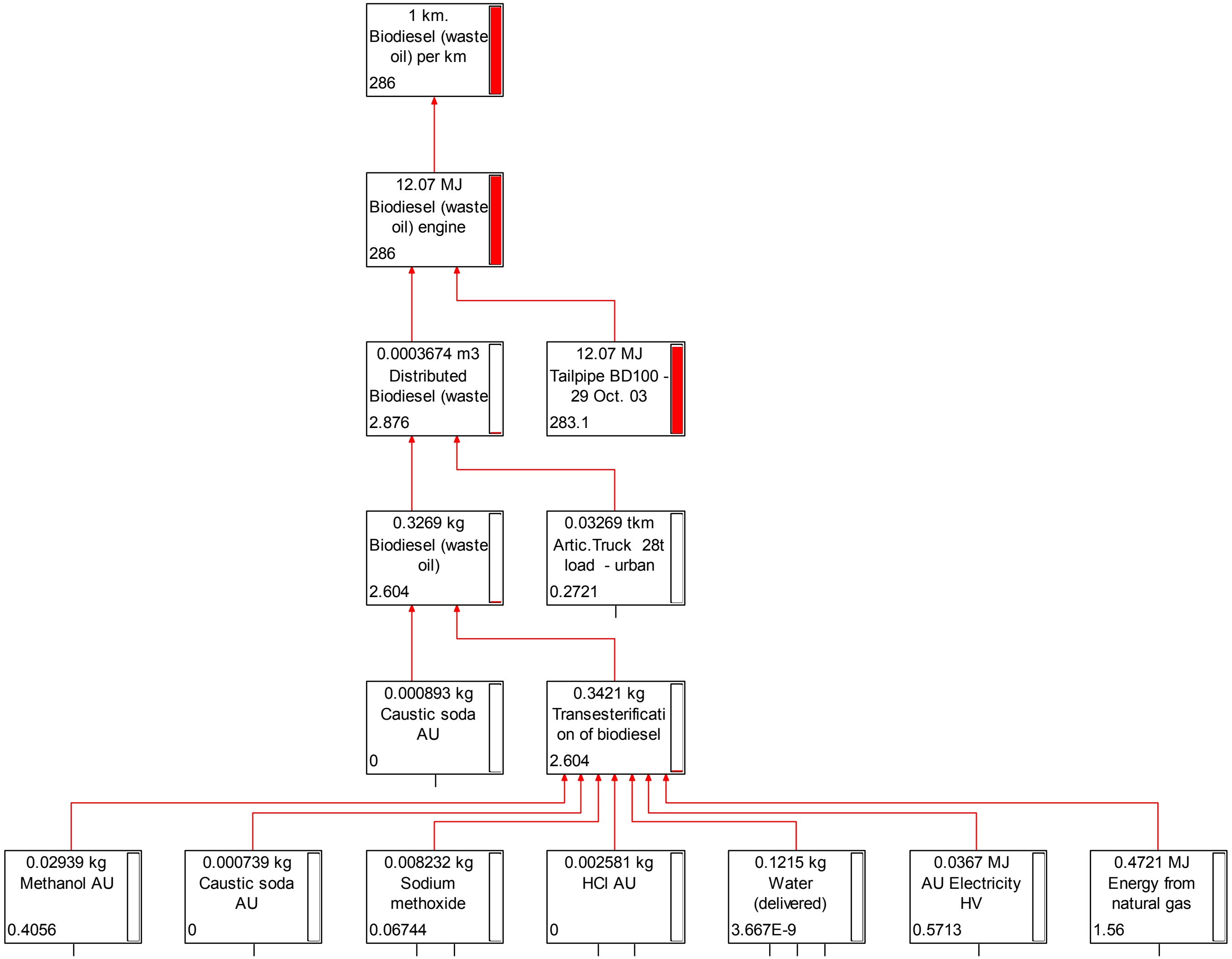


Figure A86. Biodiesel (BD100 – waste oil) emissions in mg PM per km (rigid truck)

APPENDIX VIII. UNIT HEALTH COSTS

This appendix presents a small sample of evidence on the unit health cost of various vehicle pollutant emissions. The estimates are based either on overseas estimates applied to Australia, with adjustment for Australian conditions, or on direct Australian studies.

MEASUREMENT ISSUES

The health costs of vehicle pollutants are uncertain with empirical estimates varying considerably, particularly estimates of the health cost of particulate matter (PM) emissions. Among all vehicle pollutants, recent empirical studies have generally found PM emissions to have the highest unit health cost, and also to exhibit the widest range of variation in health costs.

There are a range of issues associated with the measurement and application of unit health costs to the cost of vehicle emissions. Amoako *et al*. (2003) identify two of these issues as of particular concern: (i) multi-collinearity between pollutants, and the consequent risk of double counting health costs, and (ii) short-term versus long-term effects.

Multi-collinearity among pollutants makes it difficult to distinguish, statistically, between the health effects of different pollutants. Amoako *et al*. (2003) caution against summing the health costs for different pollutants because of the risk of double counting the health effects. Instead, Amoako *et al*. (2003) recommend using a single pollutant, PM, as a proxy for measuring total health costs. Amoako *et al*. (2003) also recommend the use of long-term health costs, in preference to short-term health costs, because the short-term health costs may understate the morbidity costs attributable to pollutant levels.[15]

Despite these concerns, the current study has attempted to cost the health impact arising from a range of pollutants and sum them to derive the total health cost. Any errors arising from double counting are likely to be small, as the health costs of PM emissions dominate the total cost estimates, contributing well over 90% of the total health cost impact.

Empirical health cost estimates

Table A73 to Table A75 present a small sample of evidence on the range of health cost estimates for vehicle pollutants. One (Watkiss 2002) based on European values adapted to Australian conditions and one (Delucchi, 2000) based on US estimates, illustrate the large range of variation in the estimates. Table A76 presents the health costs used by Coffey Geosciences (2003), in a recent report into the costs and benefits of upgrading Australian fuel standards. Table A77 reports the estimates of Beer (2002) for the unit health cost of criteria pollutant emissions derived from Australian data.

Table A73 shows estimates derived by Watkiss (2002) in a submission to the Fuel Tax Inquiry (http://fueltaxinquiry.treasury.gov.au/). The unit cost estimates are based on European values transferred to Australian conditions, controlling for population densities. The report also included a set of estimates reported by Environment Australia (2000). The major difference is the estimated cost of particulate emissions. Watkiss' estimates range from $1.24 per kg to $342 per kg depending on the location. The Environment Australia (2000) study estimated a unit cost for particulate emissions of $17.60 per kg.

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[15] Amoako *et al.*'s (2003) health cost estimates imply a unit cost of PM_{10} emissions of approximately $A 230,000–240,000 per tonne, well within the range of other estimates of the health costs of PM emissions.

Table A73. Unit Health costs of emissions – Watkiss (2002) ($A/tonne)

Emission	Area				Environment Australia (2000)
	Band 1	Band 2	Band 3	Band 4	
NO_x	1,750	1,750	260	0	1,390
CO	3	0.8	0.8	0	12
NMVOCs	850	880	180	0	1,440
SO_x	11,380	4,380	2,800	50	na
PM	341,650	93,180	93,180	1,240	17,600

Source: Watkiss (2002).
na not available.
Note Band 1 = Inner areas of larger capital cities (Sydney, Melbourne, Brisbane, Adelaide and Perth).
Band 2 = Outer areas of larger capital cities.
Band 3 = Other urban areas, including other capital cities (Canberra, Hobart and Darwin) and other urban areas.
Band 4 = Non-urban areas.

BTCE (1996), based on a small literature survey, reported the unit health costs provided in Table A74. The BTRE (as the BTCE is now known) notes that these are probably lower bound estimates, as unit costs derived for overseas cities, where higher air pollution levels are generally higher than is common in Australia, are typically much greater.

Table A74. Synthesised estimates of unit costs of environmental damage from airborne pollutants ($A/tonne)

Sector	NO_x and NMVOCs	CO	SOx	PM
Major urban				
Average	70	2	10	12,500
Range	10–70	0–20	0–450	3,000–18,000
Other				
Average	20	0	0	0
Range	0–230			

Source: BTCE (1996).

Recently, Delucchi (2000) estimated the marginal health costs of a 10% reduction in road traffic related emissions for the US. Converting to $A, and allowing for increases in prices, these figures suggest higher unit costs than Watkiss (2002) for most pollutants except for PM, for which Delucchi's costs are significantly lower at the high end of the range. For example Watkiss' estimates of the health costs per kg of PM emissions range from $1.24 to $341.65, whereas Delucchi's estimates, converted to Australian dollars range from $2.51 to $54.86.

Table A75. Marginal costs of a 10% reduction in road-traffic related emissions in the United States

	Health costs[a]			
	US$ (1991) / tonne		Equivalent $A (2003) / tonne	
Emission	Low	High	Low	High
CO	10	100	20	170
NO_x	1,500	22,080	2,600	38,220
PM10	1,450	31,690	2,510	54,860
SOx	4,400	35,280	7,620	61,080
VOC	130	1,250	230	2,160

Health costs include vehicles emissions, road dust and upstream emissions.
US$ (1991) estimates converted to $A (2003) equivalent values using the A/US exchange rate averaged across 1991 (US$0.77) and allowing for an increase in Australian consumer prices of 33% over the period June 1991 to June 2003.
Source: Delucchi (2000) and BTRE estimates.

More recently, Coffey Geosciences (2003), drawing on a large range of studies that estimated the health impact of emissions and average morbidity and mortality costs, derived unit health cost estimates for vehicular emissions in Australian capital cities. The estimates, which are listed in Table A76, range from $A12.90 per tonne of CO up to $A232,000 per tonne of PM_{10},

Table A76. Average capital city health cost saving per tonne of emission ($A/tonne)

Emission Type	Air Quality Impact	Health cost
Carbon monoxide	CO	$12.9
Oxides of Nitrogen	NO_2	$58.7
Oxides of Nitrogen	O_3	$8,500
Particulates (PM_{10})	PM_{10}	$232,000
Hydrocarbons[a]	Air toxic emissions	$2,200[a]

a. Health costs associated with hydrocarbon emissions based on US EPA (2003a) values.
Source: Coffey Geosciences (2003, table 8.5, p. 156).

Beer (2002) recently derived Australian estimates of the unit health costs of vehicular emissions of the four criteria pollutants: CO, NO_x, NMHC and PM_{10}. Beer's unit health costs are based on estimates of the annual short-term health costs of the four criteria pollutants published in the National Environment Protection Measure for Ambient Air Quality (NEPC, 1998) and estimates of the contribution of vehicles to the concentration of the criteria pollutants. Beer's (2002) estimates, reproduced in Table A77, report a cost of PM_{10} emissions ranging between $A108,000 and $A221,000 per tonne, CO emissions of between $A2 and $A9 per tonne, NO_x emissions between $A280 and $A900 per tonne and NMHC emissions between $A11,700 and $A72,500 per tonne. Beer (2002) also provides a comparison with cost estimates derived by other Australian studies, and these are included in Table A77.

Table A77. Australian valuation of health impacts of criteria pollutants ($A/tonne)

Source	Pollutant			
	CO	**HC**	**NO_x**	**PM_{10}**
Beer (2002) – Ozone included				
Upper bound	9	72,500	900	221,100
Best estimate	3	19,331	870	147,429
Lower bound	2	11,700	280	108,300
Beer (2002) – Ozone excluded	3	18,719	11	147,429
Environment Australia (2000)	12	1,440	1,385	17,600
NSW EPA (1997)	25	960	1,490	1,810
NSW EPA (1998)	na	na	68	310

na not available
Source: Beer (2002, tables 9 and 10, p. 89).

APPENDIX IX. FUEL USE, VEHICLE EMISSIONS PROJECTIONS, AND HEALTH COSTS

The fuel use and vehicle emissions projections underlying the analysis in chapter 11 are based on a range of assumptions regarding projected future vehicle use, fuel efficiency, fuel use and vehicles emissions. A summary of the major assumptions is presented here. Most of these assumptions are based on analysis presented in BTRE (2003).In the second part of this appendix the annual change in upstream and tailpipe emissions reported in chapter 11 are presented.

Aggregate Projections

Table A78 shows the BTRE (2003) base case projections of total vehicle kilometres travelled (VKT) by vehicle type for the period 2000–2001 to 2019–2020. The BTRE (2003) projected total VKT would grow by 38% between 2000–01 and 2019–20 (1.6% per annum). Total vehicle travel by passenger cars and light commercial vehicles, which, together, constitute over 90% of total VKT by all vehicles, are projected to grow by 32% (1.4% per annum) and 66% (2.6% per annum), respectively. Heavy truck (rigid trucks and articulated trucks) VKT is projected to grow by 38% (1.6% per annum) between 2000–2001 and 2019–2020.

Table A79 shows the BTRE (2003) projections of the motor vehicle stock, by vehicle type, for the period 2000–2001 to 2019–2020. The number of passenger cars and LCVs are projected to increase by 27% and 66%, respectively, between 2000–2001 and 2019–2020. The number of rigid and articulated trucks is projected to increase by 9% and 35%, respectively, between 2000–2001 and 2019–2020.

Table A80 shows the projected on-road fleet average fuel intensity, by vehicle type for the period 2000–2001 to 2019–2020. On-road average fuel consumption (the inverse of fuel intensity) of passenger cars is projected to improve by 5% between 2000–2001 and 2019–2020, reflecting lower fuel consumption rates for new vehicles. The on-road fleet average fuel consumption rates for LCVs, rigid and articulated trucks is projected to decline due to assumed increases in the average vehicle size and average vehicle load.

Table A81 shows the projected total fuel consumption in the BTRE (2003) base case, by fuel type, for the period 2000–2001 to 2019–2020. Petrol (automotive gasoline) consumption, which includes LRP, ULP and PULP, is projected to increase by 20% (0.9% per annum) between 2000–01 and 2019–20. Growth in petrol consumption is less than the projected growth in VKT travelled by petrol fuelled vehicles, reflecting assumed improvements in on-road fleet average fuel consumption. Automotive diesel use is projected to grow by 69% (2.7% per annum) reflecting assumed growth in both the average size and amount of travel by trucks. Under the base case BTRE (2003) projections, which were undertaken before the Australian Government’s announced changes to excise on alternative fuels and biofuels, ethanol use in road transport was projected to grow from around 40–45 ML to over 100 ML in 2019–2020 (4.6% per annum). As biodiesel was not widely used, BTRE (2003a) did not consider biodiesel use over the projection horizon. Table A82 reports projected road transport fuel use, shown in Table A81, in energy units.

Table A83 shows the total projected road vehicle greenhouse and criteria pollutant emissions for the BTRE (2003) base case. The BTRE (2003) base case projects total road transport end-use greenhouse emissions will increase 30% between 2000–2001 and 2019–2020. Pollutant emissions from road transport are projected to decline between 2000–01 and 2019–2020, as a result of planned future improvements in new vehicle emission rates and tighter fuel standards.

Table A78. Base case projections of national vehicle kilometres travelled by type of vehicle, 2000-01 to 2019-20 (billion kilometres)

Year	Cars	LCVs	Articulated trucks	Rigid and other trucks	Buses	Motor cycles	Total
2000–01	152.43	30.04	5.68	6.80	1.73	1.61	198.30
2001–02	155.77	31.30	5.88	6.97	1.75	1.63	203.29
2002–03	161.32	32.31	5.99	6.99	1.78	1.64	210.02
2003–04	166.67	33.26	6.22	7.17	1.80	1.66	216.77
2004–05	171.47	34.29	6.36	7.18	1.83	1.67	222.80
2005–06	174.20	35.10	6.56	7.23	1.85	1.68	226.62
2006–07	176.88	36.06	6.77	7.30	1.88	1.70	230.59
2007–08	179.52	36.87	6.97	7.34	1.90	1.71	234.31
2008–09	182.01	37.69	7.16	7.37	1.93	1.73	237.88
2009–10	184.41	38.86	7.40	7.43	1.95	1.74	241.79
2010–11	186.55	39.87	7.62	7.47	1.98	1.75	245.23
2011–12	188.62	40.88	7.84	7.49	2.00	1.76	248.59
2012–13	190.55	41.91	8.06	7.51	2.03	1.77	251.83
2013–14	192.41	42.95	8.29	7.53	2.05	1.79	255.01
2014–15	194.18	44.01	8.52	7.54	2.08	1.80	258.12
2015–16	195.85	45.07	8.75	7.55	2.10	1.81	261.12
2016–17	197.47	46.36	9.02	7.59	2.13	1.82	264.38
2017–18	199.02	47.52	9.24	7.58	2.15	1.83	267.34
2018–19	200.51	48.68	9.45	7.56	2.18	1.84	270.21
2019–20	201.95	49.84	9.66	7.52	2.21	1.85	273.02

Note ‘Passenger car’ results in all tables include 4-wheel drive passenger vehicles (‘All Terrain Wagons’), unless explicitly noted otherwise. LCV (light commercial vehicle) fleet results include the (generally) heavier 4-wheel drive vehicles primarily purchased for business uses.

Source: BTRE (2003).

Table A79. Projected vehicle stock, by vehicle type (thousand vehicles), 2000-01 to 2019-20

Year	Cars	LCVs	Rigid & other trucks	Articulated trucks	Buses	Motor cycles	All vehicles
2000–01	9995.7	1,797.9	364.0	64.3	54.0	351	12,626.9
2001–02	10181.1	1,847.3	365.8	64.6	56.0	354.1	12,868.9
2002–03	10407.5	1,905.9	372.1	65.7	56.8	357.2	13,165.1
2003–04	10649.8	1,972.9	376.6	67.1	57.6	360.2	13,484.2
2004–05	10844.8	2,022.3	377.5	67.8	58.4	363.3	13,734.1
2005–06	11016.1	2,087.9	381.1	69.1	59.2	366.3	13,979.7
2006–07	11179.6	2,149.3	383.6	70.3	60.0	369.2	14,212.1
2007–08	11330.6	2,212.4	386.2	71.5	60.8	372.1	14,433.5
2008–09	11473.6	2,275.3	388.4	72.7	61.6	375.1	14,646.6
2009–10	11617.5	2,339.2	390.4	73.9	62.4	378	14,861.5
2010–11	11745.9	2,408.4	393.1	75.3	63.2	380.5	15,066.4
2011–12	11875.7	2,476.0	395.2	76.7	64.0	383.1	15,270.6
2012–13	11999.4	2,543.6	397.0	78.0	64.8	385.6	15,468.5
2013–14	12118.2	2,611.6	398.6	79.4	65.6	388.2	15,661.6
2014–15	12232.0	2,679.3	399.8	80.8	66.4	390.7	15,849.0
2015–16	12339.4	2,745.3	400.6	82.1	67.2	393.2	16,027.8
2016–17	12442.8	2,808.8	400.8	83.4	68.0	395.6	16,199.4
2017–18	12542.5	2,869.0	400.3	84.6	68.8	398	16,363.3
2018–19	12638.6	2,924.9	399.1	85.7	69.6	400.4	16,518.3
2019–20	12731.7	2,977.2	397.2	86.7	70.7	402.8	16,666.4

Source: BTRE (2003, table 1.26, p. 32).

Table A80. Projected average on-road fleet fuel intensity, by vehicle type, 2000-01 to 2019-20 (L/100km)

Year	Cars	LCVs	Rigid & other trucks	Articulated trucks	Buses	Motor cycles
2000–01	11.44	13.34	27.43	53.48	23.48	6.00
2001–02	11.41	13.37	27.48	53.77	23.37	6.00
2002–03	11.39	13.41	27.53	54.05	23.32	6.00
2003–04	11.36	13.42	27.54	54.31	23.28	6.00
2004–05	11.34	13.46	27.61	54.54	23.26	6.00
2005–06	11.31	13.48	27.66	54.74	23.12	6.00
2006–07	11.29	13.51	27.72	54.90	23.24	6.00
2007–08	11.26	13.54	27.78	55.03	23.24	6.00
2008–09	11.23	13.57	27.85	55.16	23.26	6.00
2009–10	11.20	13.60	27.94	55.28	23.32	6.00
2010–11	11.17	13.63	28.01	55.37	23.36	6.00
2011–12	11.14	13.65	28.09	55.48	23.41	6.00
2012–13	11.10	13.68	28.18	55.58	23.46	6.00
2013–14	11.07	13.70	28.27	55.67	23.50	6.00
2014–15	11.04	13.73	28.37	55.75	23.56	6.00
2015–16	11.01	13.75	28.48	55.83	23.61	6.00
2016–17	10.98	13.78	28.60	55.93	23.71	6.00
2017–18	10.94	13.81	28.72	56.02	23.78	6.00
2018–19	10.91	13.84	28.86	56.11	23.85	6.00
2019–20	10.87	13.88	29.01	56.22	23.92	6.00

Source: BTRE (2003, table 2.18, p. 95).

Table A81. BTRE (2003) 'Base case' Projections of total motor vehicle fuel consumption by fuel type, 2000-01 to 2019-20 (ML)

Year	Petrol	ADO	LPG	NG (petrol equivalent)	Ethanol	Biodiesel	Total
2000–01	18,068.4	6,895.1	2,143.2	50.0	40.2	0	27,196.9
2001–02	18,461.1	7,163.2	2,134.6	62.6	44.9	0	27,866.4
2002–03	19,047.1	7,328.8	2,221.4	76.0	47.4	0	28,720.7
2003–04	19,599.4	7,594.0	2,285.2	89.8	49.6	0	29,618.0
2004–05	20,118.4	7,784.5	2,348.6	96.5	52.1	0	30,400.1
2005–06	20,341.5	8,008.8	2,407.4	103.5	54.7	0	30,915.9
2006–07	20,581.9	8,258.3	2,471.2	111.7	57.7	0	31,480.8
2007–08	20,797.7	8,476.7	2,532.7	120.8	60.7	0	31,988.5
2008–09	20,990.4	8,696.6	2,595.3	134.2	63.7	0	32,480.2
2009–10	21,196.8	8,968.9	2,668.1	150.6	67.1	0	33,051.5
2010–11	21,339.8	9,217.6	2,737.7	169.3	70.1	0	33,534.5
2011–12	21,469.0	9,466.6	2,808.6	191.8	73.9	0	34,009.9
2012–13	21,571.1	9,720.2	2,881.3	218.4	77.8	0	34,468.8
2013–14	21,659.4	9,980.1	2,955.6	250.3	81.6	0	34,927.0
2014–15	21,725.7	10,244.0	3,031.9	288.9	85.9	0	35,376.5
2015–16	21,762.0	10,512.7	3,109.7	334.8	90.2	0	35,809.4
2016–17	21,808.2	10,833.7	3,195.3	391.2	94.4	0	36,322.9
2017–18	21,803.2	11,108.3	3,279.0	458.5	99.6	0	36,748.5
2018–19	21,770.2	11,379.5	3,364.2	540.1	104.3	0	37,158.2
2019–20	21,712.0	11,654.7	3,447.5	639.2	109.8	0	37,563.1

Notes ADO – automotive diesel oil.
LPG – liquefied petroleum gas.
NG – natural gas.
Source*:* BTRE estimates.

Table A82. BTRE (2003) 'Base case' Projections of total motor vehicle energy consumption by fuel type, 2000-01 to 2019-20 (PJ)

Year	Petrol	ADO	LPG	NG	Ethanol	Biodiesel	Total
2000–01	617.94	266.15	55.08	1.71	0.94	0	941.82
2001–02	631.37	276.50	54.86	2.14	1.05	0	965.93
2002–03	651.41	282.89	57.09	2.60	1.11	0	995.11
2003–04	670.30	293.13	58.73	3.07	1.16	0	1,026.39
2004–05	688.05	300.48	60.36	3.30	1.22	0	1,053.40
2005–06	695.68	309.14	61.87	3.54	1.28	0	1,071.52
2006–07	703.90	318.77	63.51	3.82	1.35	0	1,091.35
2007–08	711.28	327.20	65.09	4.13	1.42	0	1,109.11
2008–09	717.87	335.69	66.70	4.59	1.49	0	1,126.35
2009–10	724.93	346.20	68.57	5.15	1.57	0	1,146.42
2010–11	729.82	355.80	70.36	5.79	1.64	0	1,163.42
2011–12	734.24	365.41	72.18	6.56	1.73	0	1,180.12
2012–13	737.73	375.20	74.05	7.47	1.82	0	1,196.27
2013–14	740.75	385.23	75.96	8.56	1.91	0	1,212.41
2014–15	743.02	395.42	77.92	9.88	2.01	0	1,228.24
2015–16	744.26	405.79	79.92	11.45	2.11	0	1,243.52
2016–17	745.84	418.18	82.12	13.38	2.21	0	1,261.73
2017–18	745.67	428.78	84.27	15.68	2.33	0	1,276.72
2018–19	744.54	439.25	86.46	18.47	2.44	0	1,291.17
2019–20	742.55	449.87	88.60	21.86	2.57	0	1,305.44

Notes ADO – automotive diesel oil.
LPG – liquefied petroleum gas.
NG – natural gas.
Source*:* BTRE estimates.

Table A83. Base case projections of end-use road transport emissions by gas type, 2000-01 to 2019-20 (kt)

Year	CO$_2$-e	CO	NO$_x$	NMVOC	SO$_x$	PM
2000–01	64,336.9	3,289.8	467.7	494.4	15.24	24.85
2001–02	66,515.1	3,208.9	467.9	481.9	14.25	24.94
2002–03	67,998.9	3,164.7	468.1	476.3	10.90	24.62
2003–04	69,296.0	3,115.5	468.4	474.1	8.65	24.48
2004–05	70,398.6	3,055.3	464.1	466.6	6.97	24.22
2005–06	71,393.3	2,962.1	455.7	454.4	5.10	23.62
2006–07	72,562.9	2,875.0	447.0	443.6	5.18	23.40
2007–08	73,616.5	2,788.7	436.1	437.6	5.24	23.24
2008–09	74,629.7	2,708.0	424.7	428.4	5.30	23.08
2009–10	75,577.6	2,631.2	414.8	420.6	5.37	22.98
2010–11	76,600.4	2,547.1	404.1	412.6	5.42	22.82
2011–12	77,573.2	2,461.2	393.1	405.3	5.48	22.65
2012–13	78,499.4	2,376.6	382.3	398.7	5.52	22.50
2013–14	79,402.7	2,295.0	372.0	392.9	5.56	22.35
2014–15	80,245.9	2,217.4	362.7	387.7	5.60	22.19
2015–16	81,061.5	2,144.0	354.1	383.1	5.63	22.05
2016–17	81,846.9	2,067.9	346.7	379.3	5.67	21.97
2017–18	82,579.2	2,026.5	340.5	377.4	5.70	21.88
2018–19	83,209.9	1,993.5	335.3	376.0	5.72	21.80
2019–20	83,723.6	1,959.4	330.9	374.8	5.73	21.71

Note CO2-e emissions include direct CO$_2$ emissions, CH$_4$ and N$_2$O emissions, weighted by the radiate forcing potential.
Sources*:* BTRE (2002 and 2003).

Passenger Car and Rigid Truck Fleet Characteristics

In order to simplify the transport impact analysis, it was assumed that all of the additional ethanol supply would be used in passenger cars and all of the additional biodiesel would be consumed in rigid trucks. This section provides some detailed information about the assumed age structure of the passenger and rigid truck fleets, and the projected average new vehicle emissions characteristics and on-road fleet average vehicle emission rates for the passenger car and rigid truck fleets.

Passenger Cars

The current (2003) passenger vehicle fleet is estimated to total 10.4 million vehicles, with 7.8 million vehicles (75 per cent of the fleet) manufactured before 2000.

Table A84 is taken from the BTRE model and shows the number of passenger vehicles (thousands) by age cohort predicted to be in the fleet in 2003, 2010 and 2020.

By 2010 the passenger vehicle fleet is projected to total 11.6 million vehicles, of which 7 million vehicles (60 per cent) are already 'on-the-road' (i.e. manufactured prior to 2003) and 4.6 million (40 per cent) will be post-2003 vehicles. Of the post-2003 vehicles, nearly 3.3 million will be produced after the Euro III standards come into effect 2005–06, and of these 1.3 million will be less than two years old (perhaps complying with Euro IV standards). The total fleet will also include over 4.7 million vehicles (40 per cent) manufactured prior to 2000.

In 2020, of the 12.7 million passenger vehicles predicted to be in the fleet, about half will be less than 10 years old and fewer than 2.7 million will come from the current (2003) passenger vehicle fleet. The total fleet will also include less than 1.3 million vehicles manufactured before 2000.

Table A85 shows the assumed pollutant emission rates for new passenger cars over the projection horizon, 2000–01 to 2019-20. Emissions of sulfur oxides fall in line with reductions in the sulfur content of all graded of petrol to 150 ppm from 1 January 2005. CO, NO_x and PM emissions for new passenger vehicles are assumed to decline in line with the mandated change in vehicle emissions standards, albeit allowing for differences between on-road emissions performance and test cycle results. The values in these tables differ from the emission rates given in Appendix VI because the values given in Appendix VI represent an average derived from the petrohol study (undertaken in 1995 - 1997), whereas the values given in Table A85 represent estimates of the emissions to be expected in new vehicles from the year 2000 onward.

Deterioration Rates

The BTRE passenger car vehicle emissions model includes estimates of the deterioration in the emissions performance of the on-road vehicle fleet, to capture the effect of vehicle tampering and sub-optimal vehicle maintenance regimes. Table A86 lists the assumed on-road vehicle emission deterioration rates for passenger cars, by vehicle age, in 2009-10.

Fleet Average On-Road Vehicle Emission Rates

Vintage specific average vehicle emission rates, combined with assumed vintage specific average vehicle utilisation are combined to derive total vehicle emissions. Table A87 shows the fleet average on-road vehicle emission rates used in the BTRE model. The reference case assumes reductions in the fleet average on-road emission rates for all major pollutants over the projection period to 2020. Over the period 2003 to 2010, of interest to the current study, fleet average on-road emission rates are projected to fall by between 20 and 60% for most pollutants. The exception is N_2O emissions, which are projected to increase slightly.

Table A84. Passenger vehicles by age (thousands)

Vehicle age	Year		
	2003	2010	2020
New	685.0	660.0	701.4
1	652.5	647.5	685.8
2	674.9	651.6	671.7
3	582.5	651.3	656.8
4	654.7	649.5	641.4
5	632.8	656.7	628.3
6	525.0	643.7	613.8
7	503.5	628.3	612.6
8	488.6	592.5	608.1
9	435.6	606.4	585.7
10	402.2	514.3	582.3
11	386.5	569.3	560.1
12	369.7	537.9	549.9
13	412.7	432.4	533.2
14	395.8	400.7	507.4
15	316.2	373.1	490.4
16	285.4	315.9	455.1
17	311.6	274.8	416.4
18	310.8	246.8	364.0
19	263.5	220.7	341.7
20	236.8	227.6	262.3
Over 20 pre-1986	881.3	526.7	96.4
Over 20 post-1986	0.0	589.6	1166.9
Total	10,407.5	11,617.5	12,731.7

Source: BTRE (2003).

Table A85. Average on-road new passenger vehicle emission rates[16], 2000-01 to 2019-20

Year	Average ULP sulfur content	Emission rates						
		SOx	N_2O	CO	NO_x	HC (Exhaust)	HC (Evap.)	PM
	(ppm)	*(g/L)*	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*
2000-01	220	0.162	0.0375	1.95	0.40	0.17	1.144	0.0145
2001-02	210	0.154	0.0375	1.80	0.35	0.17	1.139	0.0140
2002-03	200	0.147	0.0375	1.75	0.30	0.17	1.139	0.0135
2003-04	180	0.132	0.0375	1.60	0.28	0.16	0.990	0.0130
2004-05	150	0.110	0.0320	1.60	0.23	0.16	0.990	0.0125
2005-06	150	0.110	0.0288	1.60	0.21	0.15	0.891	0.0120
2006-07	150	0.110	0.0288	1.50	0.19	0.15	0.891	0.0115
2007-08	150	0.110	0.0288	1.50	0.17	0.15	0.842	0.0110
2008-09	150	0.110	0.0288	1.50	0.16	0.14	0.842	0.0105
2009-10	150	0.110	0.0288	1.50	0.15	0.13	0.842	0.0100
2010-11	150	0.110	0.0288	1.40	0.14	0.13	0.842	0.0095
2011-12	150	0.110	0.0288	1.40	0.13	0.13	0.842	0.0090
2012-13	150	0.110	0.0288	1.40	0.12	0.12	0.842	0.0085
2013-14	150	0.110	0.0288	1.40	0.12	0.12	0.842	0.0080
2014-15	150	0.110	0.0288	1.35	0.11	0.12	0.842	0.0075
2015-16	150	0.110	0.0288	1.30	0.11	0.12	0.842	0.0070
2016-17	150	0.110	0.0288	1.30	0.11	0.12	0.842	0.0065
2017-18	150	0.110	0.0288	1.30	0.11	0.11	0.842	0.0060
2018-19	150	0.110	0.0288	1.30	0.11	0.10	0.842	0.0055
2019-20	150	0.110	0.0288	1.30	0.11	0.10	0.842	0.0050

Sources: BTRE (2003) and BTRE estimates.

Table A86. Assumed on-road vehicle emissions deterioration factors (g/km) for passenger cars, 2009-10

Vehicle age	CO	NO_x	HC Exhaust	HC Evap.	PM
(years)	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*
1	0.25	0.03	0.025	0.015	0.0015
2	0.35	0.03	0.025	0.015	0.0020
3	0.35	0.03	0.025	0.015	0.0020
4	0.35	0.03	0.025	0.015	0.0020
5	0.35	0.03	0.025	0.015	0.0020
6	0.35	0.03	0.025	0.015	0.0020
7	0.50	0.05	0.030	0.015	0.0025
8	0.50	0.05	0.030	0.015	0.0025
9	0.50	0.05	0.030	0.015	0.0025
10	0.50	0.05	0.030	0.015	0.0025
11	0.50	0.05	0.030	0.015	0.0025
12	0.50	0.05	0.030	0.015	0.0025
13	0.50	0.05	0.030	0.015	0.0025
14	1.10	0.07	0.060	0.015	0.0025
15	1.10	0.07	0.060	0.015	0.0025
16	1.10	0.07	0.060	0.015	0.0025
17	1.10	0.07	0.060	0.015	0.0025
18	1.10	0.07	0.060	0.015	0.0025
19	1.10	0.07	0.060	0.015	0.0025
20	1.10	0.07	0.060	0.015	0.0025
20+ (pre-1986)	1.00	0.06	0.080	0.030	0.0020
20+ (post-1986)	1.10	0.07	0.060	0.015	0.0025

Sources: BTRE (2003) and BTRE estimates.

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[16] These values extrapolate the emissions observed during the petrohol study (Tables 29 and 30) to future years.

Table A87. Fleet average on-road emission rates – passenger cars, 2000-01 to 2019-20 (g/km)

Year	Pollutant							
	CH_4	N_2O	CO	NO_x	HC Ex.	HC Evap.	SO_x	PM
2000-01	0.149	0.045	14.96	1.30	1.10	1.52	0.034	0.046
2001-02	0.145	0.047	14.02	1.24	1.01	1.46	0.032	0.046
2002-03	0.139	0.048	13.19	1.19	0.94	1.42	0.029	0.046
2003-04	0.135	0.049	12.42	1.14	0.88	1.37	0.027	0.045
2004-05	0.131	0.050	11.68	1.09	0.82	1.32	0.024	0.045
2005-06	0.127	0.050	11.03	1.03	0.77	1.28	0.022	0.044
2006-07	0.123	0.050	10.42	0.98	0.73	1.24	0.022	0.044
2007-08	0.120	0.050	9.85	0.94	0.69	1.20	0.022	0.044
2008-09	0.117	0.050	9.34	0.89	0.65	1.17	0.022	0.044
2009-10	0.114	0.050	8.85	0.85	0.62	1.14	0.022	0.044
2010-11	0.111	0.050	8.37	0.81	0.58	1.11	0.022	0.043
2011-12	0.107	0.049	7.91	0.77	0.56	1.08	0.022	0.043
2012-13	0.104	0.049	7.48	0.72	0.53	1.06	0.022	0.042
2013-14	0.101	0.049	7.07	0.69	0.51	1.04	0.022	0.041
2014-15	0.099	0.049	6.68	0.65	0.48	1.02	0.022	0.041
2015-16	0.097	0.048	6.33	0.62	0.46	1.00	0.022	0.040
2016-17	0.095	0.048	5.97	0.59	0.45	0.99	0.021	0.039
2017-18	0.094	0.048	5.74	0.57	0.43	0.98	0.021	0.038
2018-19	0.093	0.048	5.56	0.55	0.42	0.97	0.021	0.037
2019-20	0.093	0.048	5.38	0.53	0.41	0.96	0.021	0.037

Sources: BTRE (2002) and BTRE estimates.

Rigid Trucks

The current (2003) rigid truck fleet is estimated to total 372 thousand vehicles. Table A88 is taken from the BTRE model and shows the projected number of rigid trucks, by age cohort, in the fleet in 2003, 2010 and 2020. By the year 2010, it is estimated that there will be around 25 thousand rigid trucks (6.5 per cent of the fleet) in the fleet manufactured since 2009 (the possible introduction of Euro V diesel vehicle emissions standards), a further 40 thousand vehicles (10 per cent of all rigid trucks) manufactured between 2006 and 2008, inclusive, the date of introduction of Euro IV vehicle emissions standards, and a further 40 thousand vehicles manufactured between 2003 and 2005, inclusive, the date of introduction of Euro III vehicle emissions standards.

Sulfur Content

Under the *Fuel Standard (Automotive Diesel) Determination 2001*, the sulfur content of automotive diesel is currently legislated to be not greater than 500 mg/kg, and will be limited to no greater than 50 mg/kg from 1 January 2006. European vehicle emissions standards will see the introduction of Euro V heavy vehicle standards from October 2008. Euro V heavy duty diesel vehicles will require further reductions in the sulfur content of automotive diesel to 10 mg/kg. The Motor Vehicle Environment Council is currently considering the issues associated with the introduction of Euro IV (Petrol) and Euro V (Diesel) vehicle emissions standards in Australia (MVEC, 2003). In anticipation that the introduction of these standards is largely a matter of timing, the study has assumed the standards will be legislated from 1 January 2009. This assumption has greatest consequence for sulfur oxide emissions.

Table A89 shows the assumed average sulfur content of diesel fuel used by rigid trucks, and the assumed pollutant emission rates for new rigid trucks over the period 2000–01 to 2019-20. Emissions of sulfur oxides fall in line with reductions in the sulfur content of automotive diesel. CO, NO_x and

PM emissions for new rigid trucks are assumed to decline in line with the mandated change in vehicle emissions standards, allowing for differences in on-road emissions from test cycle results.

Deterioration Rates

Like the passenger car model, the commercial vehicle models allow for deterioration in the emissions performance of the on-road vehicle fleet, to capture the effect of vehicle tampering and sub-optimal vehicle maintenance regimes. The assumed on-road deterioration rates included in the BRTE models for rigid trucks for 2010 are listed in Table A90.

Fleet Average On-Road Vehicle Emission Rates

Fleet average vehicle emissions rates are equal to the vehicle travel weighted average of vintage specific average vehicle emission rates. Table A91 shows the fleet average on-road vehicle emission rates used in the BTRE model. The reference case assumes reductions in the fleet average on-road emission rates for all major pollutants over the projection period to 2020. Over the period 2003 to 2010, of interest to the current study, fleet average on-road emission rates are projected to fall by between 20 and 40% for most pollutant. The exceptions are N_2O emissions, which are projected to increase slightly, evaporative emissions, which are projected to increase slightly due to changes in the assumed fuel mix, and SO_x emissions, which fall by over 95% due to reductions in the sulfur content of automotive diesel.

Table A88. Rigid trucks by age (thousands)

Vehicle age	Year		
	2003	2010	2020
New	16.3	12.7	9.1
1	11.8	12.8	9.8
2	12.0	13.0	10.5
3	13.4	12.9	11.1
4	13.0	13.8	11.7
5	11.2	11.1	12.0
6	9.3	14.5	12.3
7	9.4	15.9	12.3
8	11.0	11.4	12.5
9	9.4	11.5	12.8
10	9.3	12.8	12.1
11	9.4	12.4	12.1
12	9.3	10.6	12.2
13	12.6	8.7	11.9
14	14.9	8.7	12.7
15	12.5	10.1	10.0
16	9.6	8.5	12.9
17	12.1	8.3	13.8
18	17.8	8.2	9.8
19	13.3	7.9	9.6
20	8.8	10.6	10.5
Over 20 pre-1986	125.6	114.3	42.3
Over 20 post-1986	0.0	39.6	113.3
Total	372.1	390.4	397.2

Source: BTRE (2003).

Table A89. Diesel fuel sulfur content and average on-road new rigid truck emission rates, 2000-01 to 2019-20

Year	Average diesel sulfur content	Emission rates						
		SOx	N_2O	CO	NO_x	HC (Exhaust)	HC (Evap.)	PM
	(ppm)	*(g/L)*	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*
2000-01	900	0.7614	0.080	3.0	5.00	0.70	2.388	0.3500
2001-02	800	0.6768	0.080	3.0	5.00	0.65	2.376	0.3000
2002-03	500	0.4230	0.080	2.5	4.80	0.60	2.376	0.2000
2003-04	300	0.2538	0.080	2.0	4.50	0.60	2.376	0.1500
2004-05	200	0.1692	0.080	1.8	4.00	0.60	2.376	0.1000
2005-06	50	0.0423	0.080	1.7	3.50	0.50	2.376	0.1000
2006-07	50	0.0423	0.080	1.5	3.50	0.45	2.376	0.0500
2007-08	50	0.0423	0.080	1.5	3.00	0.45	2.376	0.0400
2008-09	30	0.0254	0.080	1.4	2.00	0.45	2.376	0.0300
2009-10	10	0.0085	0.080	1.4	2.00	0.40	2.376	0.0200
2010-11	10	0.0085	0.100	1.4	2.00	0.40	2.376	0.0195
2011-12	10	0.0085	0.100	1.3	2.00	0.40	2.376	0.0190
2012-13	10	0.0085	0.100	1.3	2.00	0.40	2.376	0.0185
2013-14	10	0.0085	0.100	1.3	2.00	0.40	2.376	0.0180
2014-15	10	0.0085	0.100	1.3	2.00	0.40	2.376	0.0175
2015-16	10	0.0085	0.100	1.3	2.00	0.30	2.376	0.0170
2016-17	10	0.0085	0.100	1.2	2.00	0.30	2.376	0.0165
2017-18	10	0.0085	0.100	1.2	2.00	0.30	2.376	0.0160
2018-19	10	0.0085	0.100	1.2	2.00	0.30	2.376	0.0155
2019-20	10	0.0085	0.100	1.2	2.00	0.30	2.376	0.0150

Sources: BTRE (2003) and BTRE estimates.

Table A90. Assumed on-road vehicle emissions deterioration factors (g/km) for rigid trucks, 2009-10

Vehicle age	CO	NO_x	HC Ex.	HC Evap.	PM
(years)	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*	*(g/km)*
1	0.2	0.03	0.025	0.015	0.01
2	0.2	0.03	0.025	0.015	0.01
3	0.2	0.03	0.025	0.015	0.01
4	0.2	0.03	0.025	0.015	0.01
5	0.2	0.03	0.025	0.015	0.01
6	0.2	0.03	0.025	0.015	0.01
7	0.1	0.05	0.020	0.015	0.01
8	0.1	0.05	0.020	0.015	0.01
9	0.1	0.05	0.020	0.015	0.01
10	0.1	0.05	0.020	0.015	0.01
11	0.1	0.05	0.020	0.015	0.01
12	0.1	0.05	0.020	0.015	0.01
13	0.1	0.05	0.020	0.015	0.01
14	0.1	0.07	0.020	0.015	0.01
15	0.1	0.07	0.020	0.015	0.01
16	0.1	0.07	0.020	0.015	0.01
17	0.1	0.07	0.020	0.015	0.01
18	0.1	0.07	0.020	0.015	0.01
19	0.1	0.07	0.020	0.015	0.01
20	0.1	0.07	0.020	0.015	0.01
20+ (pre-1986)	0.1	0.06	0.030	0.030	0
20+ (post-1986)	0.1	0.07	0.020	0.015	0.01

Sources: BTRE (2003) and BTRE estimates.

Table A91. Fleet average on-road emission rates – rigid trucks, 2000-01 to 2019-20 (g/km)

Year	Pollutant							
	CH_4	N_2O	CO	NO_x	HC Ex.	HC Evap.	SO_x	PM
2000-01	0.056	0.045	5.51	7.41	1.932	2.72	0.3884	0.66
2001-02	0.052	0.049	5.38	7.24	1.838	2.68	0.3474	0.64
2002-03	0.048	0.053	5.20	7.05	1.731	2.63	0.2187	0.61
2003-04	0.044	0.057	4.99	6.84	1.623	2.60	0.1318	0.58
2004-05	0.041	0.060	4.80	6.65	1.535	2.56	0.0883	0.54
2005-06	0.038	0.064	4.62	6.43	1.453	2.54	0.0229	0.51
2006-07	0.035	0.067	4.45	6.22	1.379	2.52	0.0228	0.49
2007-08	0.033	0.070	4.30	6.01	1.313	2.50	0.0227	0.47
2008-09	0.032	0.073	4.15	5.76	1.252	2.49	0.0139	0.45
2009-10	0.030	0.075	4.02	5.50	1.195	2.48	0.0051	0.44
2010-11	0.029	0.078	3.90	5.25	1.144	2.46	0.0051	0.42
2011-12	0.028	0.082	3.79	5.01	1.098	2.45	0.0052	0.41
2012-13	0.027	0.085	3.70	4.79	1.058	2.44	0.0052	0.39
2013-14	0.027	0.088	3.61	4.58	1.023	2.43	0.0052	0.38
2014-15	0.026	0.091	3.54	4.39	0.993	2.42	0.0053	0.37
2015-16	0.026	0.094	3.47	4.21	0.958	2.41	0.0053	0.36
2016-17	0.025	0.096	3.41	4.04	0.918	2.40	0.0053	0.35
2017-18	0.025	0.099	3.36	3.89	0.893	2.39	0.0054	0.34
2018-19	0.025	0.101	3.33	3.76	0.872	2.38	0.0054	0.33
2019-20	0.026	0.103	3.31	3.65	0.852	2.38	0.0054	0.32

Sources: BTRE (2002) and BTRE estimates.

Emissions Projections

This section provides the annual estimates of the change in vehicle emissions, resulting from the increase in total biofuel supply required to obtain 350 ML in 2010, described in chapter 11. The annual change in total health costs resulting from the change in biofuel supply between the reference case and the 350 ML biofuel case are presented in the final three tables (Table A100, Table A101 and Table A102). All tables in this section provide annual results for the period 2000-01 to 2019-20. For all cases, there is no change in biofuel supply and use until 2003-04 (for ethanol), hence there is no change in emissions until that date.

Tailpipe Emissions

Table A92 shows the change in tailpipe greenhouse and major pollutant emissions, between 2000–2001 and 2019–2020, from increased ethanol consumption. Table A93 shows the projected change in tailpipe emission from increased biodiesel consumption between the reference case and the 350 ML biofuels consumption case. The total change in projected tailpipe emissions are shown in Table A94.

Table A92. Change in projected tailpipe emissions from increased ethanol consumption, 2000–01 to 2019–20 (tonnes)

Year	Greenhouse	Pollutants			
	CO_2-e	CO	NO_x	VOCs	PM
2000–01	0	0	0	0	0
2001–02	0	0	0	0	0
2002–03	0	0	0	0	0
2003–04	-15,801	-3,689	69	120	-0.06
2004–05	-34,941	-7,768	148	265	-0.13
2005–06	-64,715	-13,741	263	488	-0.23
2006–07	-104,358	-21,177	409	780	-0.36
2007–08	-157,056	-30,482	592	1,204	-0.54
2008–09	-226,449	-42,148	822	1,722	-0.76
2009–10	-311,313	-55,531	1,087	2,360	-1.03
2010–11	-311,322	-53,193	1,045	2,358	-1.01
2011–12	-311,331	-50,868	1,002	2,360	-0.99
2012–13	-311,340	-48,652	961	2,369	-0.96
2013–14	-311,348	-46,560	922	2,381	-0.94
2014–15	-311,355	-44,588	886	2,395	-0.91
2015–16	-311,361	-42,782	854	2,413	-0.89
2016–17	-311,364	-40,884	822	2,438	-0.87
2017–18	-311,366	-39,911	799	2,448	-0.85
2018–19	-311,366	-39,189	780	2,467	-0.84
2019–20	-311,367	-38,469	763	2,491	-0.82

Table A93. Change in projected tailpipe emissions from increased biodiesel consumption, 2000-01 to 2019–20 (tonnes)

Year	Greenhouse	Pollutants			
	CO_2-e	CO	NO_x	VOCs	PM
2000–01	0	0.0	0.0	0.0	0.00
2001–02	0	0.0	0.0	0.0	0.00
2002–03	0	0.0	0.0	0.0	0.00
2003–04	0	0.0	0.0	0.0	0.00
2004–05	0	0.0	0.0	0.0	0.00
2005–06	0	0.0	0.0	0.0	0.00
2006–07	-12,795	-37.3	14.7	-5.0	-1.51
2007–08	-25,548	-74.4	28.5	-9.6	-2.90
2008–09	-50,984	-145.1	54.7	-18.4	-5.58
2009–10	-75,011	-207.6	173.3	-18.6	-5.85
2010–11	-74,819	-203.5	165.3	-17.9	-5.63
2011–12	-74,594	-199.9	157.8	-17.2	-5.44
2012–13	-74,353	-197.0	151.0	-16.6	-5.26
2013–14	-74,097	-194.9	144.5	-16.1	-5.10
2014–15	-73,823	-193.5	138.5	-15.7	-4.95
2015–16	-73,542	-191.6	132.7	-15.2	-4.82
2016–17	-73,215	-190.4	127.3	-14.6	-4.68
2017–18	-72,879	-190.3	122.7	-14.2	-4.56
2018–19	-72,513	-192.0	118.6	-13.9	-4.45
2019–20	-72,115	-194.3	115.0	-13.6	-4.34

Table A94. Change in projected tailpipe emissions from increased biofuel consumption, 2000-01 to 2019-20 (tonnes)

Year	Greenhouse	Pollutants			
	CO_2-e	CO	NO_x	VOCs	PM
2000–01	0	0	0	0	0
2001–02	0	0	0	0	0
2002–03	0	0	0	0	0
2003–04	-15,801	-3,689	69	120	-0.06
2004–05	-34,941	-7,768	148	265	-0.13
2005–06	-64,715	-13,741	263	488	-0.23
2006–07	-117,153	-21,214	424	775	-1.87
2007–08	-182,604	-30,556	620	1,194	-3.44
2008–09	-277,433	-42,294	876	1,703	-6.35
2009–10	-386,324	-55,738	1,260	2,341	-6.88
2010–11	-386,142	-53,397	1,210	2,340	-6.64
2011–12	-385,925	-51,068	1,160	2,343	-6.42
2012–13	-385,693	-48,849	1,112	2,352	-6.22
2013–14	-385,445	-46,755	1,067	2,365	-6.04
2014–15	-385,178	-44,782	1,025	2,380	-5.86
2015–16	-384,903	-42,974	986	2,398	-5.70
2016–17	-384,579	-41,074	949	2,423	-5.55
2017–18	-384,245	-40,101	921	2,433	-5.41
2018–19	-383,878	-39,381	898	2,453	-5.29
2019–20	-383,482	-38,663	878	2,477	-5.16

Upstream Emissions

Table A95, Table A96, Table A97, Table A98 and Table A99 show the change in upstream emissions from production of 30 ML of additional biodiesel in 2010 and the change in upstream emissions resulting from the reduction in diesel production.

Table A95. Change in upstream emissions from additional ethanol production, 2000-01 to 2019-20 (tonnes)

Year	Greenhouse	Pollutants			
	CO_2-e	CO	NO_x	VOCs	PM
2000–01	0	0.0	0.0	0.0	0.00
2001–02	0	0.0	0.0	0.0	0.00
2002–03	0	0.0	0.0	0.0	0.00
2003–04	4,553	119.7	17.5	3.2	0.22
2004–05	10,765	283.0	41.3	7.6	0.52
2005–06	19,136	503.1	73.5	13.6	0.92
2006–07	34,761	703.0	123.2	20.0	4.75
2007–08	66,548	733.5	205.7	24.7	18.05
2008–09	110,486	875.6	336.5	38.5	36.04
2009–10	182,889	1,798.5	666.4	111.2	63.08
2010–11	182,889	1,798.5	666.4	111.2	63.08
2011–12	182,889	1,798.5	666.4	111.2	63.08
2012–13	182,889	1,798.5	666.4	111.2	63.08
2013–14	182,889	1,798.5	666.4	111.2	63.08
2014–15	182,889	1,798.5	666.4	111.2	63.08
2015–16	182,889	1,798.5	666.4	111.2	63.08
2016–17	182,889	1,798.5	666.4	111.2	63.08
2017–18	182,889	1,798.5	666.4	111.2	63.08
2018–19	182,889	1,798.5	666.4	111.2	63.08
2019–20	182,889	1,798.5	666.4	111.2	63.08

Table A96. Change in upstream emissions from reduction in domestic petrol production, 2000-01 to 2019-20 (tonnes)

Year	Greenhouse	Pollutants			
	CO_2-e	CO	NO_x	VOCs	PM
2000–01	0	0.0	0.0	0.0	0.0
2001–02	0	0.0	0.0	0.0	0.0
2002–03	0	0.0	0.0	0.0	0.0
2003–04	-2,819	-3.8	-19.9	-7.8	-0.60
2004–05	-6,664	-9.0	-47.0	-18.5	-1.41
2005–06	-11,847	-15.9	-83.5	-32.8	-2.51
2006–07	-18,768	-25.2	-132.3	-52.0	-3.98
2007–08	-27,943	-37.5	-197.0	-77.5	-5.93
2008–09	-40,037	-53.8	-282.3	-111.0	-8.49
2009–10	-55,909	-75.1	-394.1	-155.0	-11.86
2010–11	-55,909	-75.1	-394.1	-155.0	-11.86
2011–12	-55,909	-75.1	-394.1	-155.0	-11.86
2012–13	-55,909	-75.1	-394.1	-155.0	-11.86
2013–14	-55,909	-75.1	-394.1	-155.0	-11.86
2014–15	-55,909	-75.1	-394.1	-155.0	-11.86
2015–16	-55,909	-75.1	-394.1	-155.0	-11.86
2016–17	-55,909	-75.1	-394.1	-155.0	-11.86
2017–18	-55,909	-75.1	-394.1	-155.0	-11.86
2018–19	-55,909	-75.1	-394.1	-155.0	-11.86
2019–20	-55,909	-75.1	-394.1	-155.0	-11.86

Table A97. Change in upstream emissions from additional biodiesel production, 2000-01 to 2019-20 (tonnes)

Year	Greenhouse	Pollutants			
	CO_2-e	CO	NO_x	VOCs	PM
2000–01	0	0.0	0.0	0.0	0.00
2001–02	0	0.0	0.0	0.0	0.00
2002–03	0	0.0	0.0	0.0	0.00
2003–04	0	0.0	0.0	0.0	0.00
2004–05	0	0.0	0.0	0.0	0.00
2005–06	0	0.0	0.0	0.0	0.00
2006–07	1,304	2.4	4.7	0.75	0.03
2007–08	2,609	4.9	9.5	1.50	0.07
2008–09	5,217	9.7	19.0	3.00	0.13
2009–10	7,826	14.6	28.5	4.50	0.23
2010–11	7,826	14.6	28.5	4.50	0.23
2011–12	7,826	14.6	28.5	4.50	0.23
2012–13	7,826	14.6	28.5	4.50	0.23
2013–14	7,826	14.6	28.5	4.50	0.23
2014–15	7,826	14.6	28.5	4.50	0.23
2015–16	7,826	14.6	28.5	4.50	0.23
2016–17	7,826	14.6	28.5	4.50	0.23
2017–18	7,826	14.6	28.5	4.50	0.23
2018–19	7,826	14.6	28.5	4.50	0.23
2019–20	7,826	14.6	28.5	4.50	0.23

Table A98. Change in upstream emissions from reduction in diesel production, 2000-01 to 2019-20 (tonnes)

Year	Greenhouse	Pollutants			
	CO_2-e	CO	NO_x	VOCs	PM
2000–01	0	0.0	0.0	0.0	0.00
2001–02	0	0.0	0.0	0.0	0.00
2002–03	0	0.0	0.0	0.0	0.00
2003–04	0	0.0	0.0	0.0	0.00
2004–05	0	0.0	0.0	0.0	0.00
2005–06	0	0.0	0.0	0.0	0.00
2006–07	-2,526	-4.7	-17.3	-6.4	-0.24
2007–08	-5,051	-9.3	-34.5	-12.7	-0.48
2008–09	-10,102	-18.7	-69.1	-25.5	-0.95
2009–10	-16,760	-30.5	-115.5	-39.2	-1.50
2010–11	-16,760	-30.5	-115.5	-39.2	-1.50
2011–12	-16,760	-30.5	-115.5	-39.2	-1.50
2012–13	-16,760	-30.5	-115.5	-39.2	-1.50
2013–14	-16,760	-30.5	-115.5	-39.2	-1.50
2014–15	-16,760	-30.5	-115.5	-39.2	-1.50
2015–16	-16,760	-30.5	-115.5	-39.2	-1.50
2016–17	-16,760	-30.5	-115.5	-39.2	-1.50
2017–18	-16,760	-30.5	-115.5	-39.2	-1.50
2018–19	-16,760	-30.5	-115.5	-39.2	-1.50
2019–20	-16,760	-30.5	-115.5	-39.2	-1.50

Table A99. Total change in exbodied emissions due to 350 ML biofuel consumption by 2010, 2001–2020 (tonnes)

Year	Greenhouse	Pollutants			
	CO_2-e	CO	NO_x	VOCs	PM
2000–01	0	0	0	0	0.00
2001–02	0	0	0	0	0.00
2002–03	0	0	0	0	0.00
2003–04	-14,067	-3,573	67	116	-0.44
2004–05	-30,841	-7,494	142	254	-1.03
2005–06	-57,426	-13,254	253	468	-1.82
2006–07	-102,381	-20,539	402	738	-1.30
2007–08	-146,441	-29,864	604	1,131	8.27
2008–09	-211,869	-41,481	880	1,608	20.39
2009–10	-268,278	-54,031	1,445	2,263	43.05
2010–11	-268,095	-51,689	1,396	2,261	43.28
2011–12	-267,879	-49,360	1,345	2,264	43.50
2012–13	-267,647	-47,141	1,297	2,274	43.70
2013–14	-267,398	-45,047	1,252	2,286	43.89
2014–15	-267,131	-43,074	1,210	2,301	44.06
2015–16	-266,857	-41,266	1,172	2,319	44.22
2016–17	-266,533	-39,367	1,134	2,345	44.38
2017–18	-266,199	-38,394	1,107	2,355	44.51
2018–19	-265,832	-37,674	1,083	2,374	44.64
2019–20	-265,436	-36,956	1,063	2,399	44.77

Table A100. Health cost impact of increased ethanol supply, 2000-01 to 2019-20

Year	Extra ethanol	Health cost impact					
		CO	NO_x	VOCs	PM	Total	Average cost
	(ML)			($ million)			*(c/L)*
2000–01	0.0	0.00	0.00	0.00	0.00	0.00	na
2001–02	0.0	0.00	0.00	0.00	0.00	0.00	na
2002–03	0.0	0.00	0.00	0.00	0.00	0.00	na
2003–04	10.3	-0.01	0.03	-0.05	-0.16	-0.19	-1.85
2004–05	24.4	-0.01	0.05	-0.12	-0.37	-0.45	-1.83
2005–06	43.4	-0.02	0.09	-0.20	-0.66	-0.79	-1.82
2006–07	68.8	-0.03	0.14	-0.31	-0.94	-1.14	-1.66
2007–08	102.5	-0.04	0.19	-0.45	-1.05	-1.35	-1.31
2008–09	146.8	-0.06	0.26	-0.62	-1.17	-1.59	-1.09
2009–10	205.0	-0.08	0.34	-0.82	-1.22	-1.79	-0.87
2010–11	205.0	-0.07	0.31	-0.79	-1.22	-1.78	-0.87
2011–12	205.0	-0.07	0.28	-0.77	-1.22	-1.78	-0.87
2012–13	205.0	-0.07	0.25	-0.74	-1.21	-1.78	-0.87
2013–14	205.0	-0.07	0.22	-0.72	-1.21	-1.77	-0.87
2014–15	205.0	-0.06	0.19	-0.70	-1.21	-1.77	-0.86
2015–16	205.0	-0.06	0.17	-0.68	-1.20	-1.77	-0.86
2016–17	205.0	-0.06	0.15	-0.67	-1.20	-1.77	-0.86
2017–18	205.0	-0.06	0.13	-0.66	-1.20	-1.78	-0.87
2018–19	205.0	-0.06	0.12	-0.65	-1.19	-1.78	-0.87
2019–20	205.0	-0.05	0.11	-0.65	-1.19	-1.78	-0.87

Table A101. Health cost impact of increased biodiesel supply, 2000-01 to 2019-20

Year	Extra biodiesel	Health cost impact					
		CO	NO_x	VOCs	PM	Total	Average cost
	(ML)			($ million)			*(c/L)*
2000–01	0.0	0	0	0	0	0.00	na
2001–02	0.0	0	0	0	0	0.00	na
2002–03	0.0	0	0	0	0	0.00	na
2003–04	0.0	0	0	0	0	0.00	na
2004–05	0.0	0	0	0	0	0.00	na
2005–06	0.0	0	0	0	0	0.00	na
2006–07	5.0	0.00	0.00	-0.01	-0.36	-0.37	-7.46
2007–08	10.0	0.00	-0.01	-0.01	-0.70	-0.72	-7.24
2008–09	20.0	0.00	-0.02	-0.02	-1.36	-1.41	-7.03
2009–10	30.0	0.00	0.04	-0.03	-1.54	-1.54	-5.13
2010–11	30.0	0.00	0.03	-0.03	-1.50	-1.50	-5.01
2011–12	30.0	0.00	0.02	-0.03	-1.46	-1.47	-4.90
2012–13	30.0	0.00	0.02	-0.03	-1.42	-1.44	-4.80
2013–14	30.0	0.00	0.01	-0.03	-1.39	-1.41	-4.71
2014–15	30.0	0.00	0.00	-0.03	-1.36	-1.39	-4.63
2015–16	30.0	0.00	0.00	-0.03	-1.33	-1.37	-4.55
2016–17	30.0	0.00	-0.01	-0.03	-1.31	-1.34	-4.47
2017–18	30.0	0.00	-0.01	-0.03	-1.28	-1.32	-4.41
2018–19	30.0	0.00	-0.01	-0.03	-1.26	-1.30	-4.35
2019–20	30.0	0.00	-0.02	-0.03	-1.24	-1.28	-4.28

Table A102. Health cost impact of increased biofuel supply, 2000-01 to 2019-20

Year	Extra biofuel	Health cost impact					
		CO	NO_x	VOCs	PM	Total	Average cost
	(ML)	($ million)					*(c/L)*
2000–01	0.0	0.00	0.00	0.00	0.00	0.00	na
2001–02	0.0	0.00	0.00	0.00	0.00	0.00	na
2002–03	0.0	0.00	0.00	0.00	0.00	0.00	na
2003–04	0.0	-0.01	0.03	-0.05	-0.16	-0.19	-1.85
2004–05	23.0	-0.01	0.05	-0.12	-0.37	-0.45	-1.83
2005–06	42.6	-0.02	0.09	-0.20	-0.66	-0.79	-1.82
2006–07	73.7	-0.03	0.13	-0.32	-1.30	-1.52	-2.05
2007–08	113.4	-0.04	0.18	-0.46	-1.75	-2.07	-1.84
2008–09	169.1	-0.06	0.24	-0.65	-2.53	-3.00	-1.80
2009–10	235.0	-0.08	0.37	-0.85	-2.77	-3.33	-1.42
2010–11	235.0	-0.08	0.34	-0.82	-2.72	-3.28	-1.40
2011–12	235.0	-0.07	0.30	-0.80	-2.68	-3.25	-1.38
2012–13	235.0	-0.07	0.26	-0.77	-2.64	-3.22	-1.37
2013–14	235.0	-0.07	0.23	-0.75	-2.60	-3.19	-1.36
2014–15	235.0	-0.06	0.20	-0.73	-2.57	-3.16	-1.35
2015–16	235.0	-0.06	0.17	-0.71	-2.53	-3.14	-1.34
2016–17	235.0	-0.06	0.14	-0.70	-2.50	-3.11	-1.33
2017–18	235.0	-0.06	0.12	-0.69	-2.48	-3.10	-1.32
2018–19	235.0	-0.06	0.10	-0.68	-2.45	-3.09	-1.31
2019–20	235.0	-0.05	0.09	-0.68	-2.43	-3.07	-1.31

APPENDIX X. UNCERTAINTY EMISSION FACTORS AND POLLUTION COSTS PER KM

CSIRO used the software @Risk (Palisade Corporation) to perform an uncertainty analysis and examine the probability distribution associated with the percentage difference between the biofuels and reference/base fuels.

The uncertainty in the life cycle analysis (LCA) parts is parameterised using triangular probability distributions.

The results for stochastic emission factors are shown in Figures A87 to A122.

Ethanol emissions per km

Greenhouse gases

Figures A87 to A91 depict the probability distributions of the percentage change in the exbodied GHG emissions per km resulting from E10 compared to ULP.

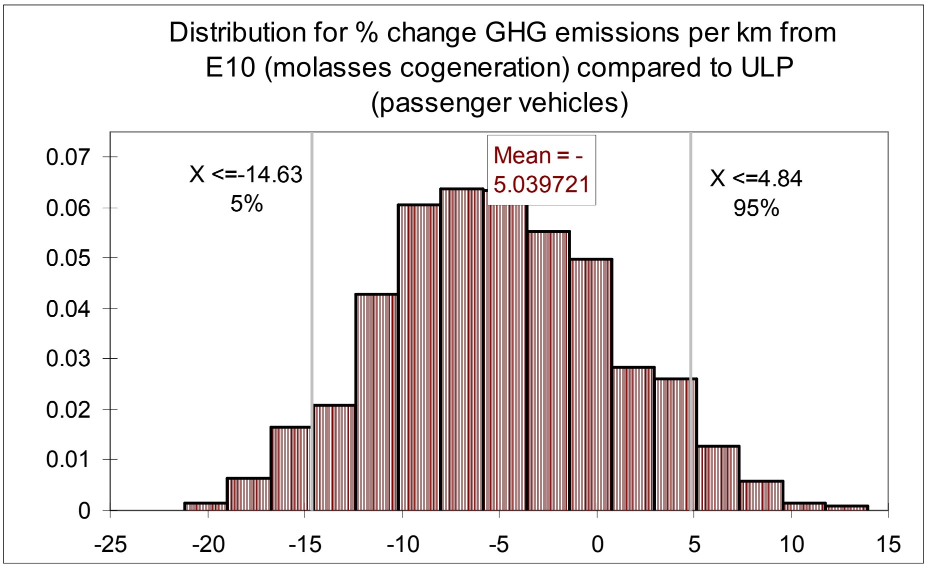


Figure A87. Distribution of percentage change in greenhouse gas emissions from E10 (molasses cogeneration) compared to ULP (passenger vehicle)

The chart above indicates that the average reduction is 5%, but there is still 20.73% probability of obtaining an increase on GHG emissions per km with E10 (molasses cogeneration).

The variation in the upstream and tailpipe processes leads to the conclusion that when compared to ULP, there is 90% probability that the GHG emissions of E10 lie between –14.6% and +4.8%. There could be a reduction of up to 14.6% or an increase up to 4.8%.

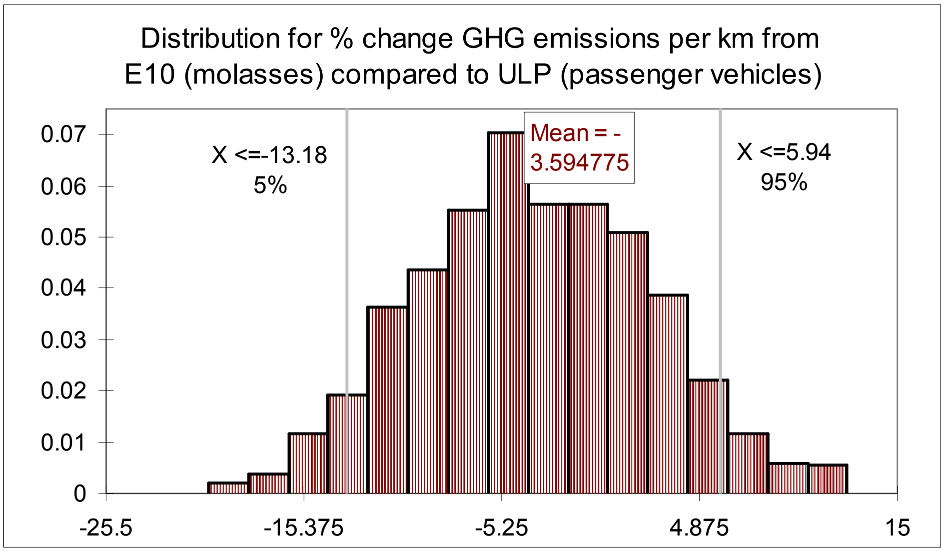


Figure A88. Distribution of percentage change in greenhouse gas emissions from E10 (molasses) compared to ULP (passenger vehicle)

When molasses is used for ethanol production, the average percentage reduction in exbodied GHG emissions is 3.6% (Figure A88). There is still a 28.3% probability of obtaining increased GHG emissions with E10 (molasses).

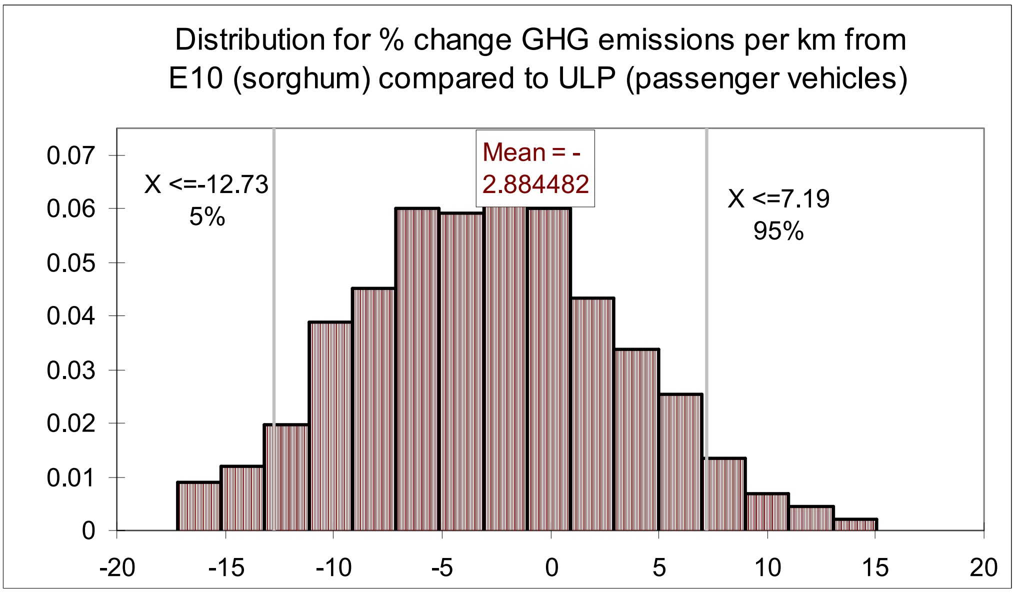


Figure A89. Distribution of percentage change in greenhouse gas emissions from E10 (sorghum) compared to ULP (passenger vehicle)

The average percentage reduction in exbodied GHG emissions from combustion of E10 (sorghum) compared to ULP is 2.9% (Figure A89). The probability of an increase in GHG emissions with E10 (sorghum) is 31.9%.

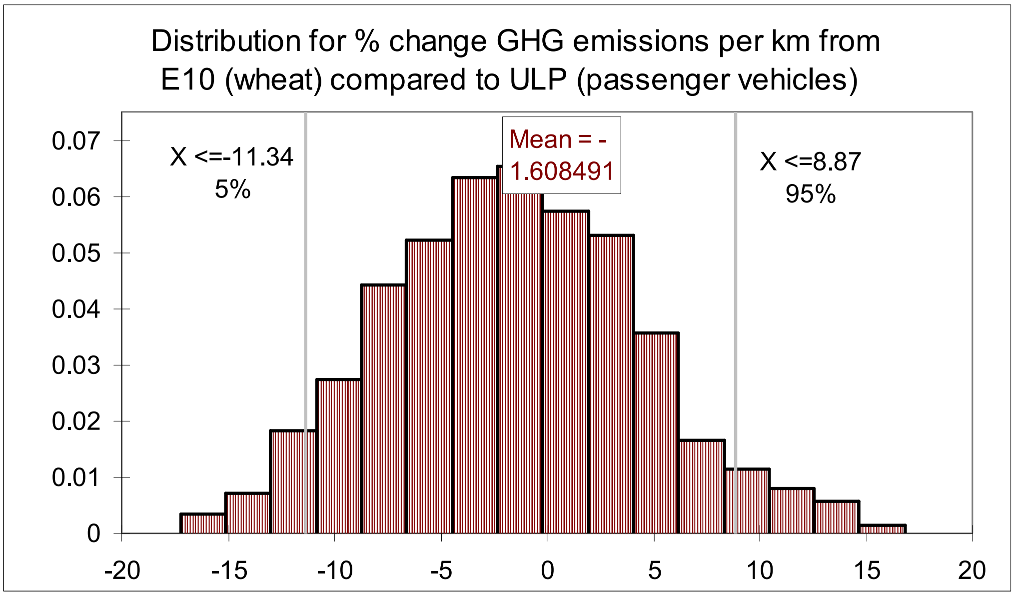


Figure A90. Distribution of percentage change in greenhouse gas emissions from E10 (wheat) compared to ULP (passenger vehicle)

Figure A90 presents the distribution of the percentage changes in exbodied GHG emissions per km when ULP is replaced with E10 sourced from wheat. This indicates that in 90% of cases the changes are anywhere between -11.3% to 8.9% and in 39.17% of situations there is likely to be an increase in GHG emissions with E10 (wheat).

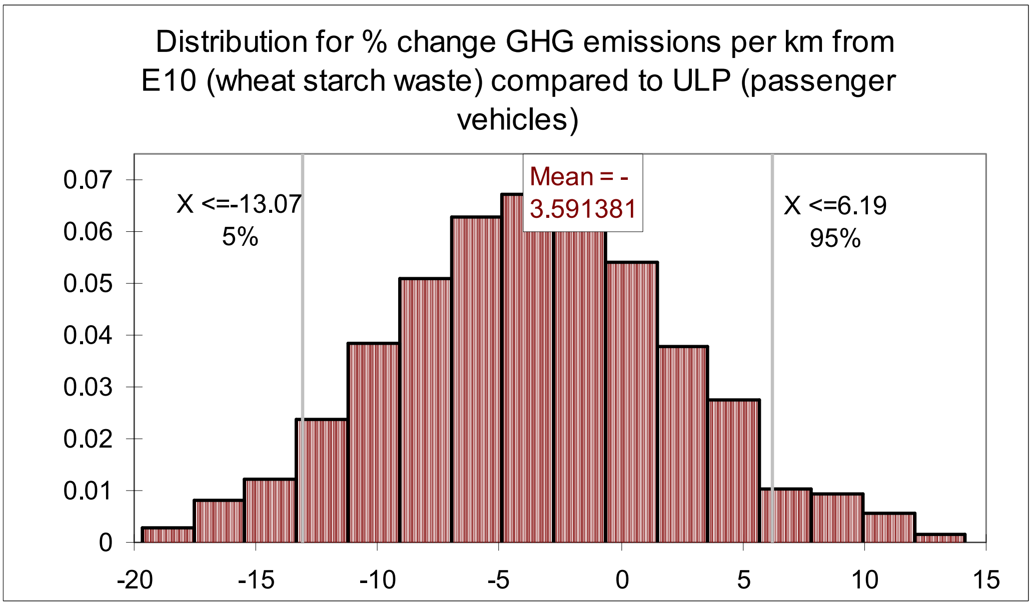


Figure A91. Distribution of percentage change in greenhouse gas emissions from E10 (wheat starch waste) compared to ULP (passenger vehicle)

The combustion of starch waste E10 leads to an average reduction of exbodied GHG emissions of 3.6% when compared to ULP (Figure A91). There is also a 27.2% probability of obtaining an increase in GHG emissions with E10 (wheat starch waste) rather than a reduction.

Air pollutants

Similar distributions are obtained for criteria air pollutants. Figures A92 to A96 present the distributions for percentage changes in CO emissions for each of the ethanol feedstocks. The average reductions are 16% (molasses - Figure A92and Figure A93, and wheat - Figure A95) and 21% (sorghum - Figure A94 and starch waste - Figure A96). The average values obtained in the simulation are lower than the best estimates considered in the distribution due to the skewness of distributions.

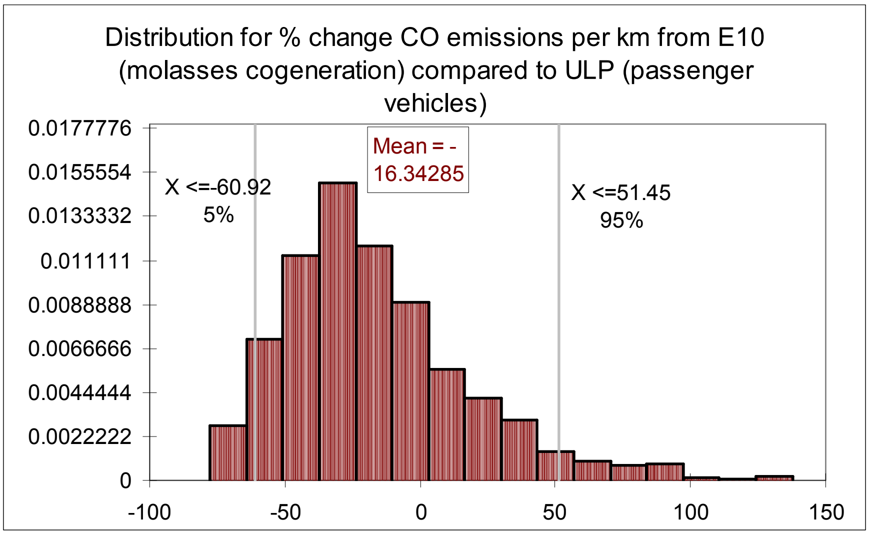


Figure A92. Distribution of percentage change in CO emissions from E10 (molasses cogeneration) compared to ULP (passenger vehicle)

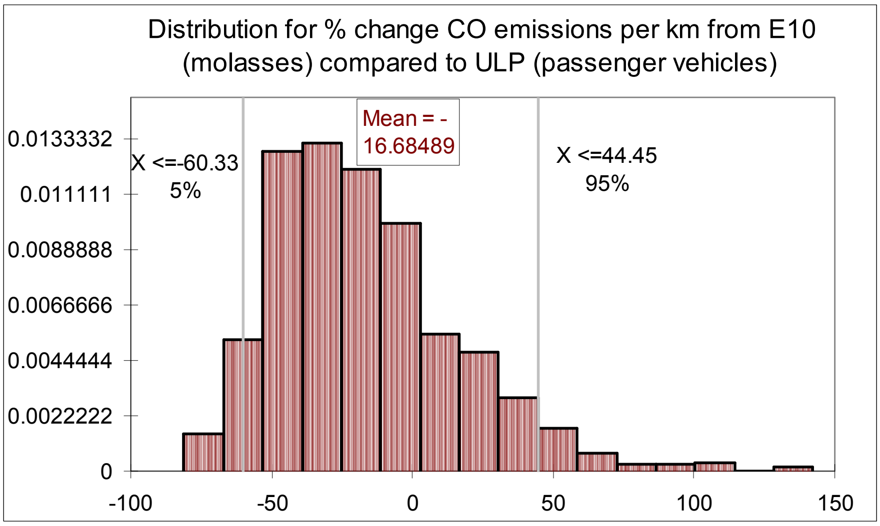


Figure A93. Distribution of percentage change in CO emissions from E10 (molasses) compared to ULP (passenger vehicle)

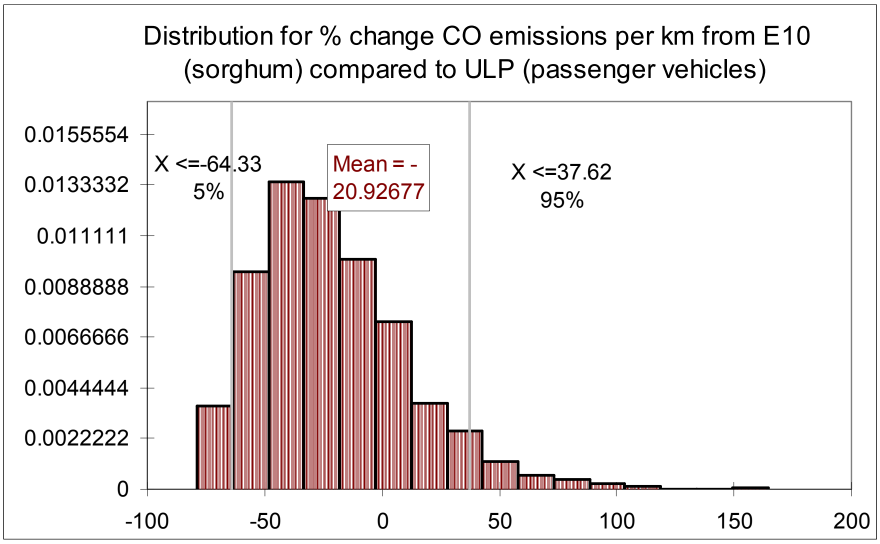


Figure A94. Distribution of percentage change in CO emissions from E10 (sorghum) compared to ULP (passenger vehicle)

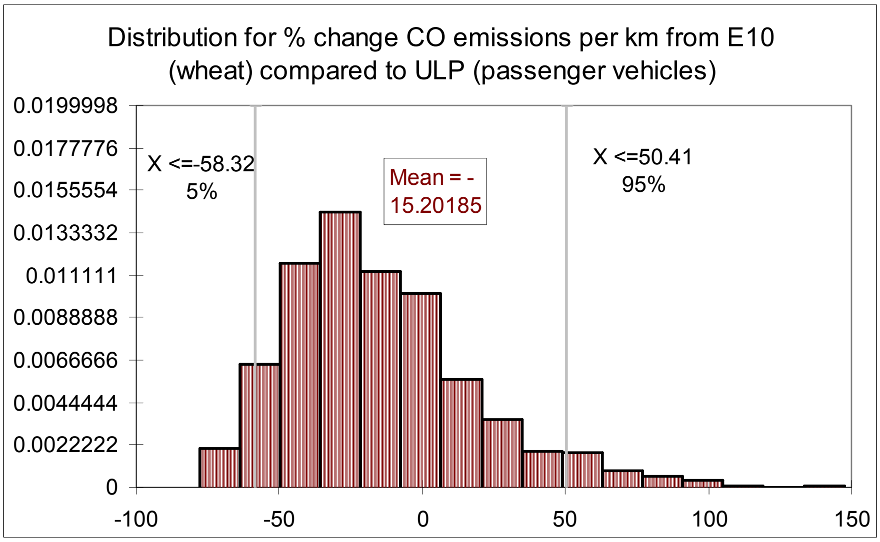


Figure A95. Distribution of percentage change in CO emissions from E10 (wheat) compared to ULP (passenger vehicle)

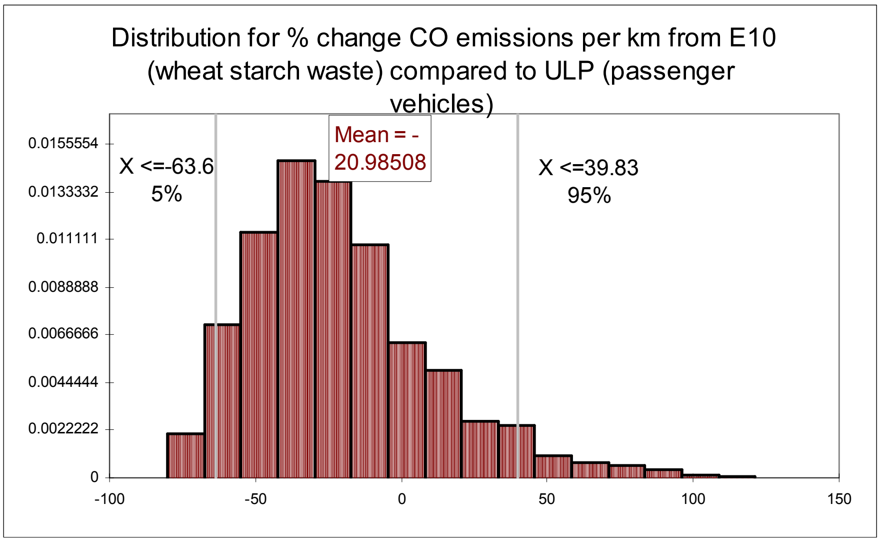


Figure A96. **Distribution of percentage change in CO emissions from E10 (wheat starch waste) compared to ULP (passenger vehicle)**

The large uncertainties in CO emissions and the asymmetry of the distributions lead also to non-negligible probabilities of obtaining increases, rather than reductions (between 25 and 28%).

Figures A97 to A101 present the distributions for percentage changes in NO_x emissions when E10 replaces ULP.

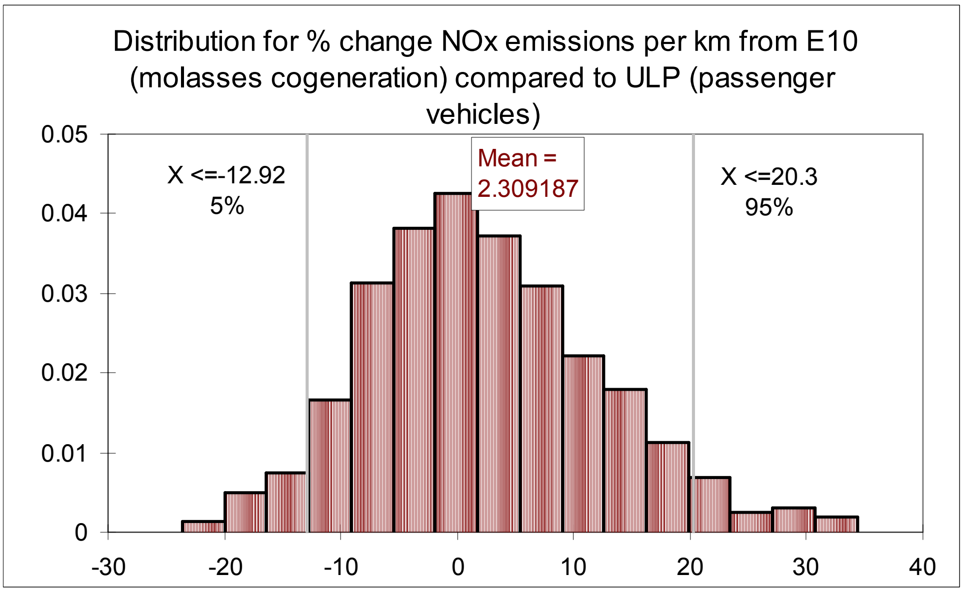


Figure A97. **Distribution of percentage change in NO_x emissions from E10 (molasses cogeneration) compared to ULP (passenger vehicle)**

The increases in NO_x emissions resulting from E10 (molasses cogeneration) combustion compared to ULP vary largely around the average of 2.3% (Figure A97). There is also a 44.5% probability of obtaining a reduction in NO_x emissions from E10 (molasses cogeneration).

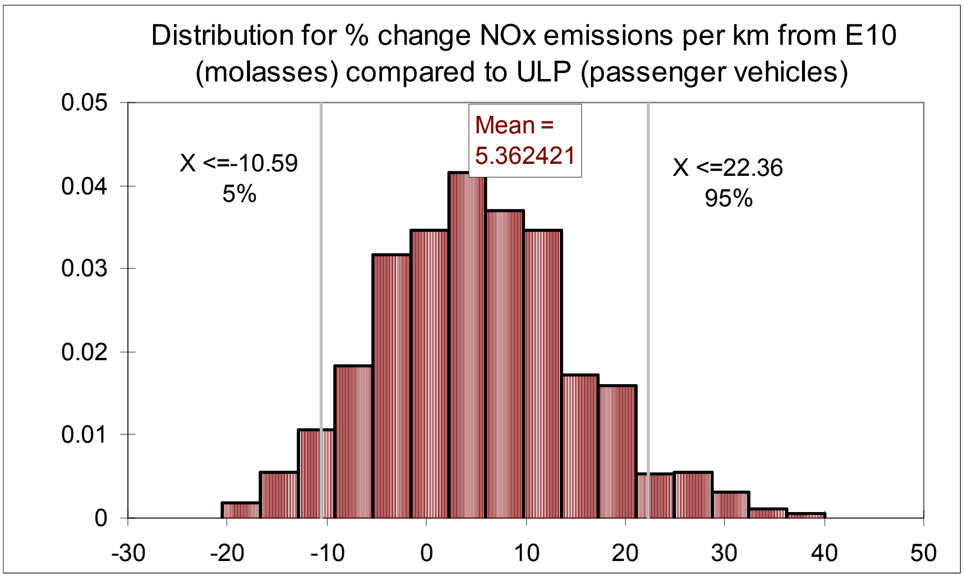


Figure A98. **Distribution of percentage change in NO_x emissions from E10 (molasses) compared to ULP (passenger vehicle)**

Larger increases (on average 5.4%) in NO_x emissions occur when cogeneration is not used for ethanol production from molasses. There is also a 30.1% probability of obtaining a reduction in NO_x emissions from E10 (molasses) - Figure A98.

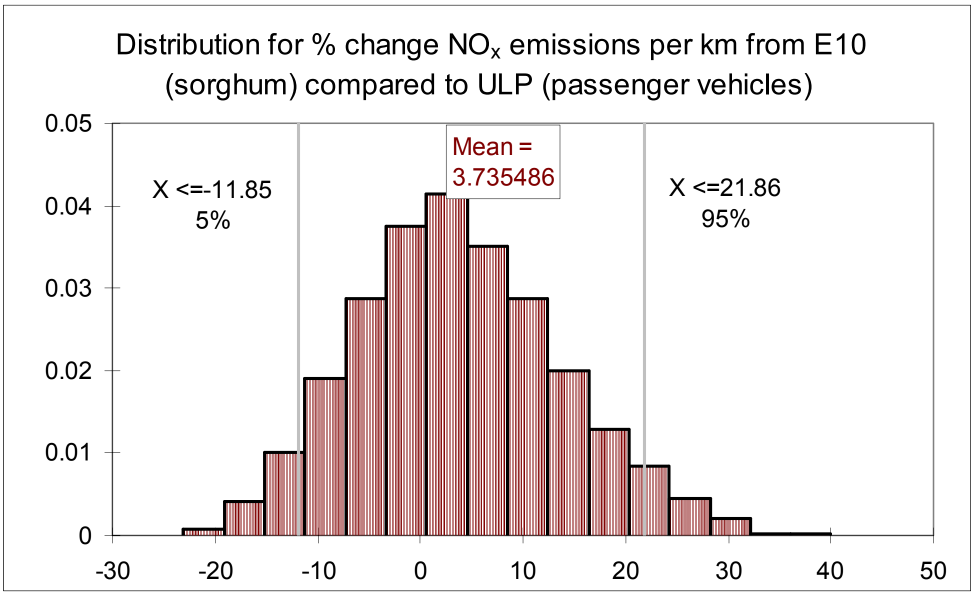


Figure A99. **Distribution of percentage change in NO_x emissions from E10 (sorghum) compared to ULP (passenger vehicle)**

When sorghum is used as a feedstock for ethanol production, the NO_x emissions are, on average 3.7% higher than emissions from ULP (Figure A99). The probability of obtaining a reduction in NO_x emissions from E10 (sorghum) is 37.5%.

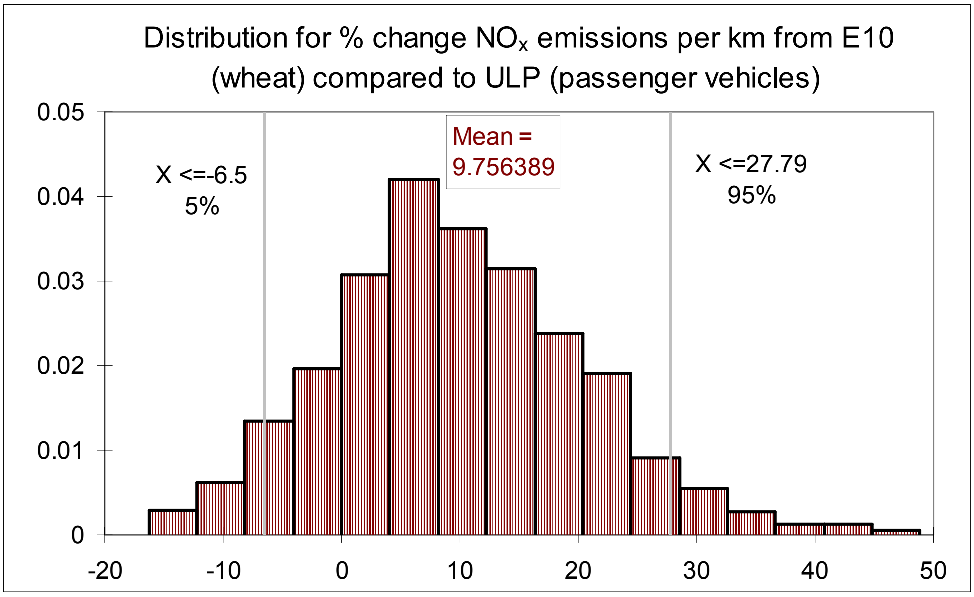


Figure A100. Distribution of percentage change in NO$_x$ emissions from E10 (wheat) compared to ULP (passenger vehicle)

The largest increase in NO$_x$ emissions per km from E10 combustion occurs when ethanol is produced from wheat, leading to a 9.7% increase in emissions per km. There is a small probability —17.2% - of obtaining a reduction in NO$_x$ emissions from E10 (wheat) compared to ULP - Figure A100.

If the ethanol source is starch waste, the increase in NO$_x$ emissions per km of E10 is only 3.37%, with 36.2% probability of actually obtaining a reduction in NO$_x$ emissions - Figure A101.

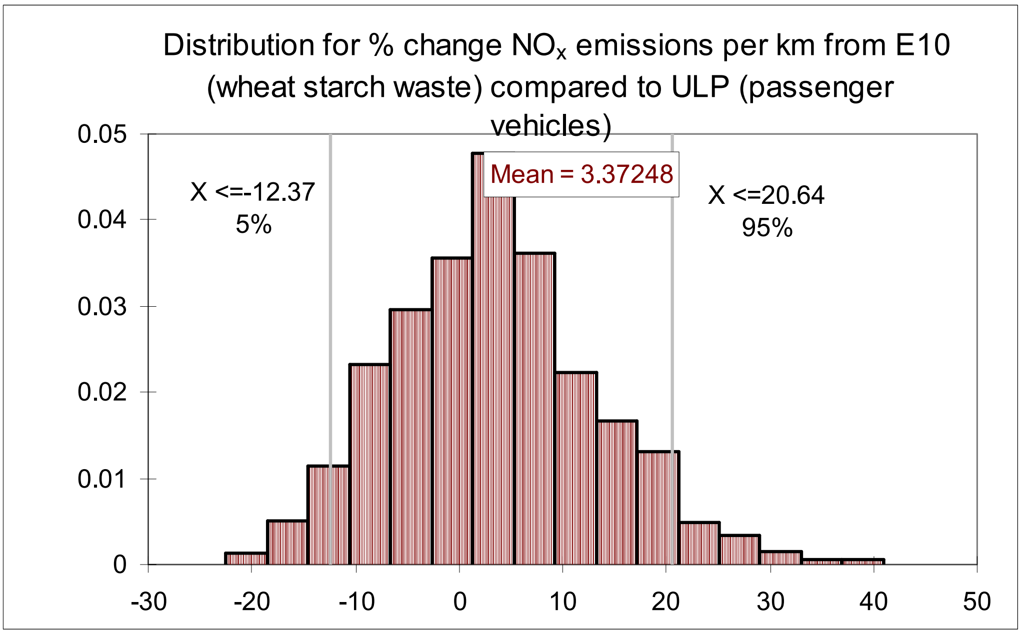
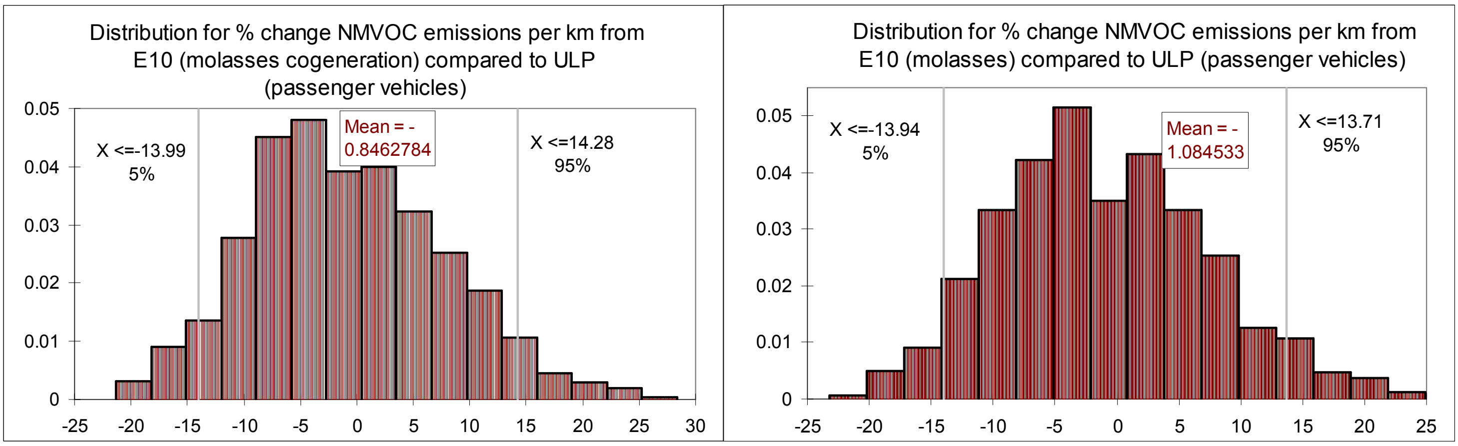


Figure A101. Distribution of percentage change in NO$_x$ emissions from E10 (wheat starch waste) compared to ULP (passenger vehicle)

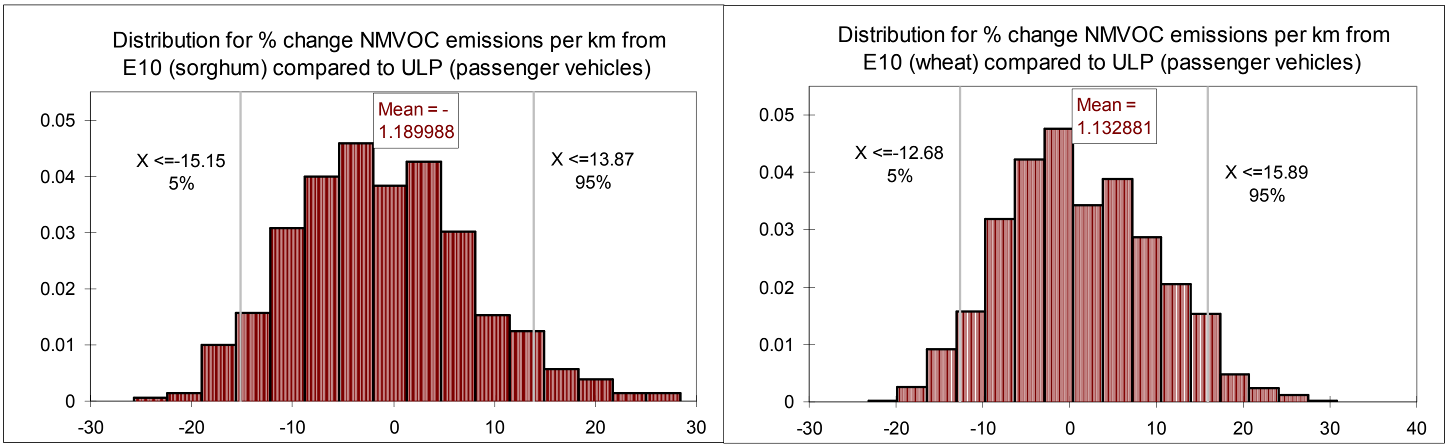
Figure A102(a to e) summarises the percentage changes in NMVOC emissions per km from E10. Regardless of the feedstock, the changes are very small (either positive or negative) and due to the high variance no statistical conclusion can be drawn about the direction of the changes.

The following probabilities of NMVOC emissions increases/reductions illustrate that there is equal probability for increases and reductions of NMVOC emissions from combustion of E10:
a 43.06% probability for increases in NMVOC emissions from E10 (molasses cogeneration);
a 43.65% probability for increases in NMVOC emissions from E10 (molasses);
a 42.98% probability for increases in NMVOC emissions from E10 (sorghum);
a 49.11% probability for reductions in NMVOC emissions from E10 (wheat);
a 42.48% probability for increases in NMVOC emissions from E10 (wheat starch waste).



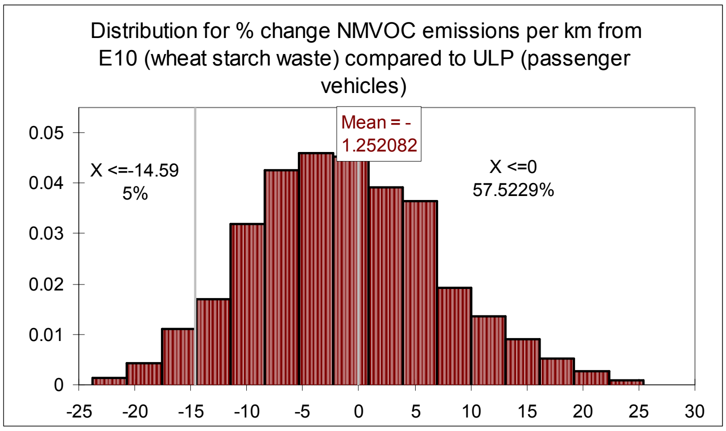
a) E10 molasses cogeneration

b) E10 molasses



c) E10 sorghum

d) E10 wheat



e) E10 wheat starch waste

Figure A102. Distribution of percentage change in NMVOC emissions from E10 compared to ULP (passenger vehicle)

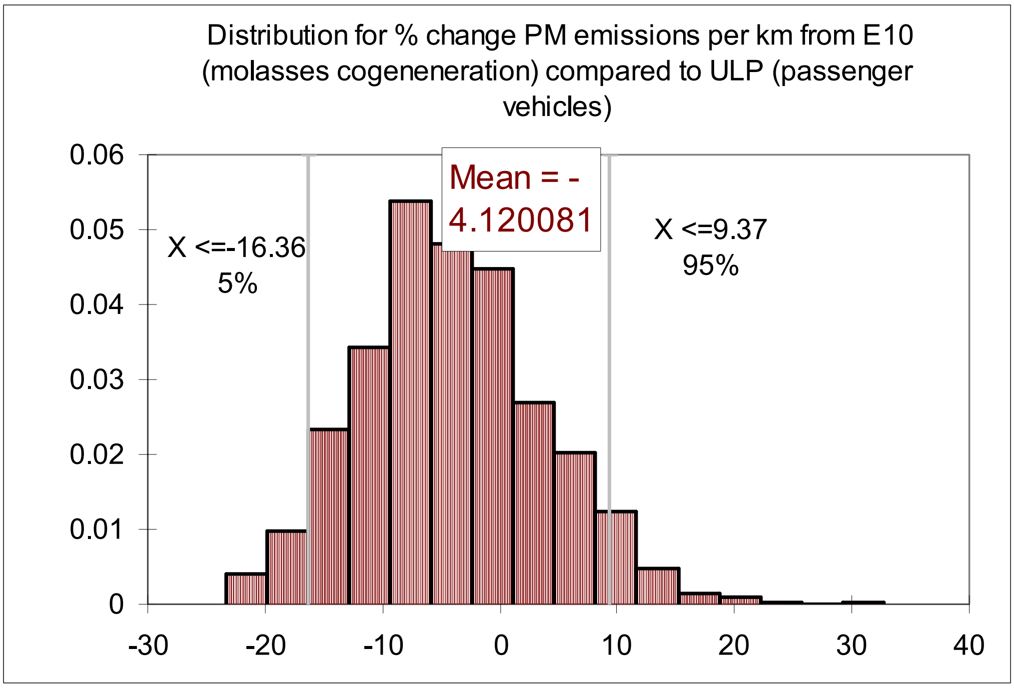


Figure A103. Distribution of percentage change in PM emissions from E10 (molasses cogeneration) compared to ULP (passenger vehicle)

Only ethanol from molasses, using cogeneration, leads to savings in PM emissions per km (Figure A103). There is also a probability of 27.2% of obtaining increases in PM when E10 from molasses produced using cogeneration replaces ULP.

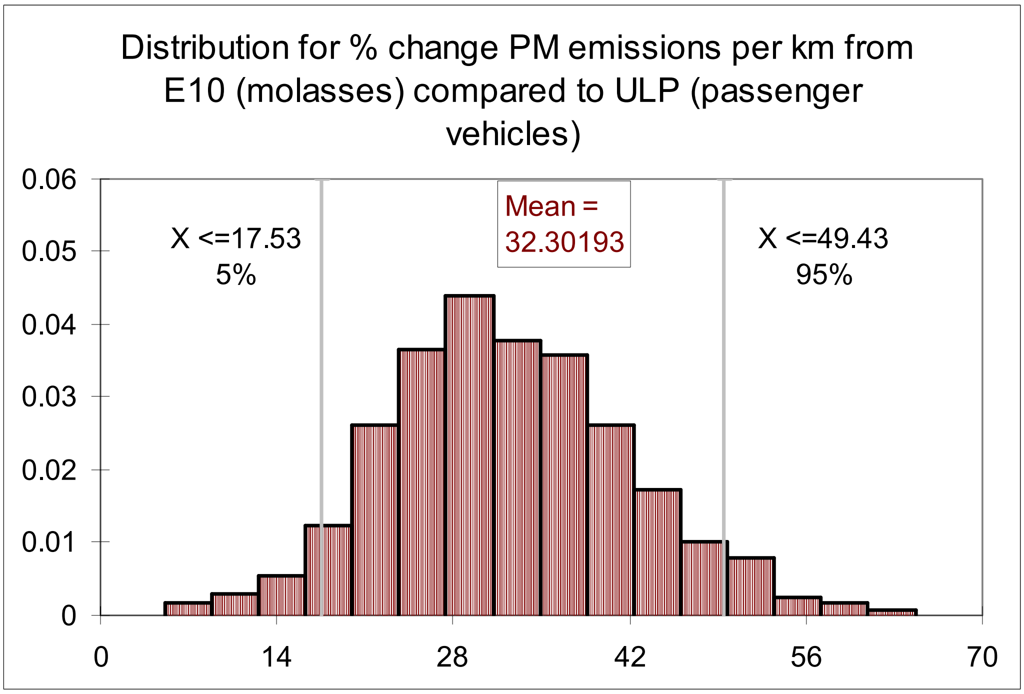


Figure A104. Distribution of percentage change in PM emissions from E10 (molasses) compared to ULP (passenger vehicle)

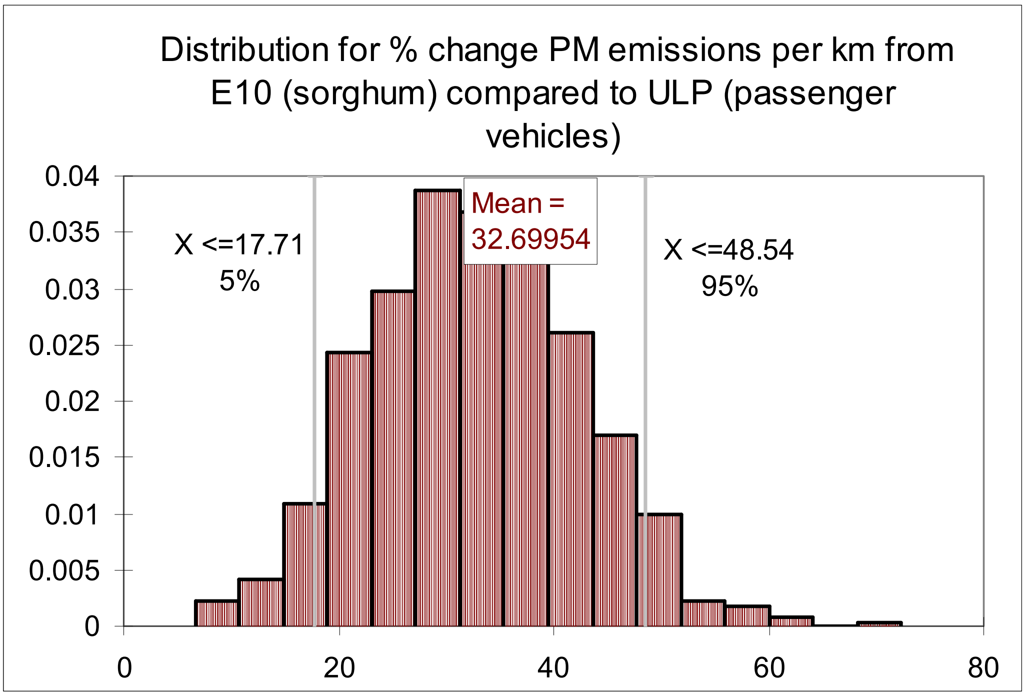


Figure A105. **Distribution of percentage change in PM emissions from E10 (sorghum) compared to ULP (passenger vehicle)**

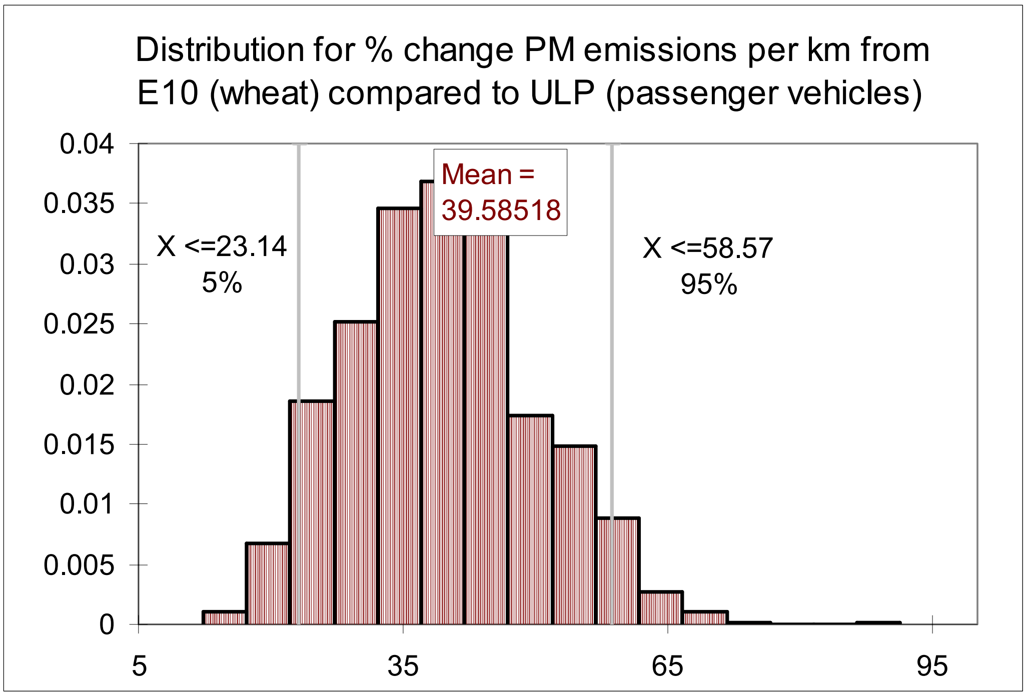


Figure A106. **Distribution of percentage change in PM emissions from E10 (wheat) compared to ULP (passenger vehicle)**

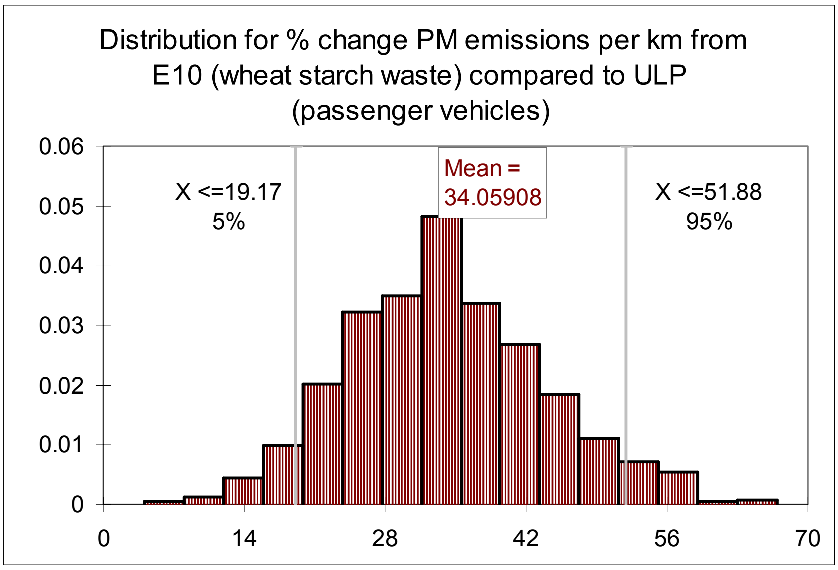


Figure A107. Distribution of percentage change in PM emissions from E10 (wheat starch waste) compared to ULP (passenger vehicle)

Biodiesel emissions per km

Greenhouse gas emissions

Similar distributions have been determined for GHG emissions and air pollutants from each of the biodiesel fuels (pure or in 20% and 5% blend) compared to the three base diesel fuels. For space limitation reasons we present here only the changes in emissions obtained for pure biodiesel compared to LSD. Upon request, all distributions for all fuels can be made available.

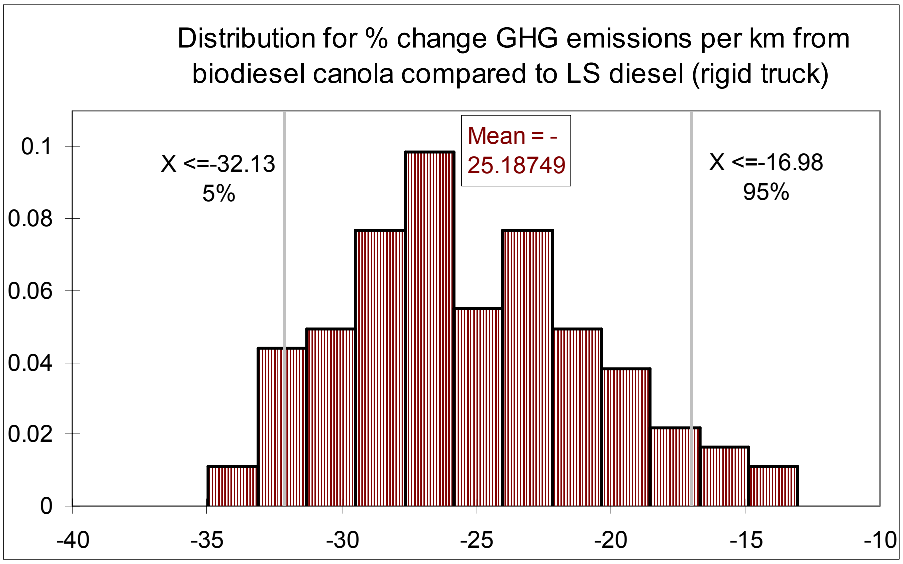


Figure A108. Distribution of percentage change in GHG emissions per km from BD100 (canola) compared to LSD (rigid truck)

Figures A108 to A110 show the distribution of exbodied GHG emissions per km when LSD is replaced with pure biodiesel (BD100) from the three feedstocks: canola oil, tallow, and waste oil.

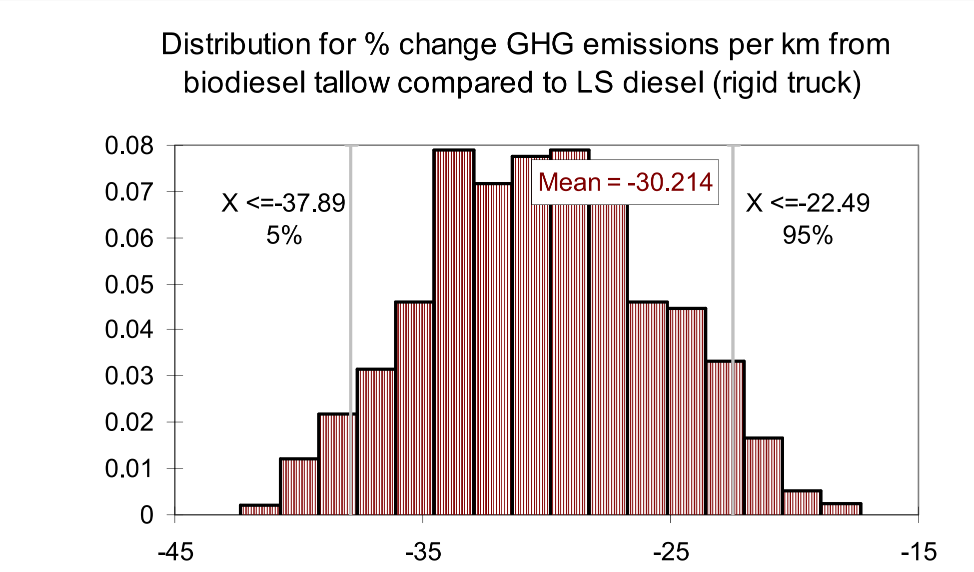


Figure A109. Distribution of percentage change in GHG emissions per km from BD100 (tallow) compared to LSD (rigid truck)

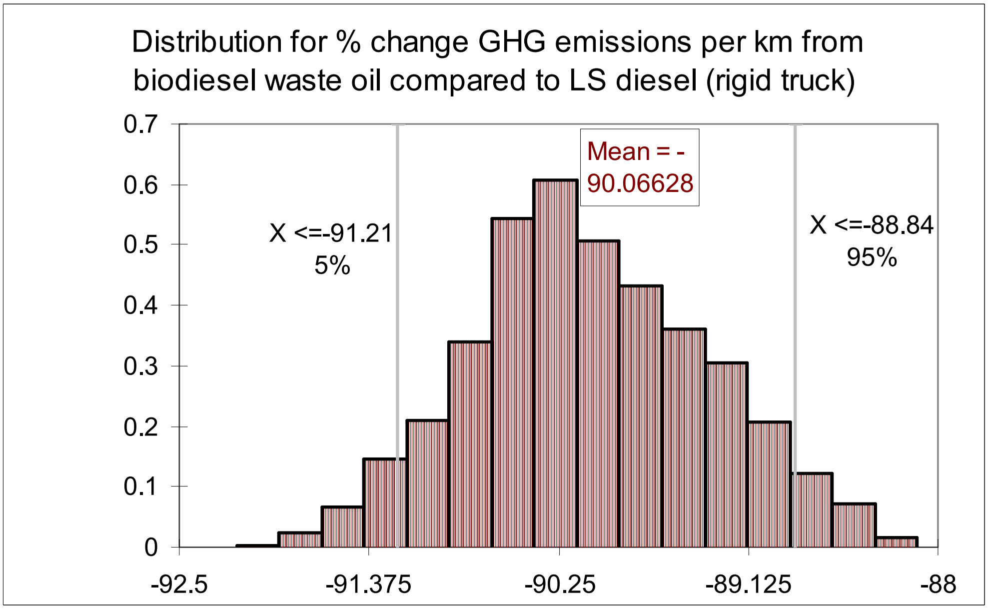


Figure A110. Distribution of percentage change in GHG emissions per km from BD100 (waste oil) compared to LSD (rigid truck)

Significant reductions in GHG emissions per km are obtained when biodiesel replaces LSD: between 25 and 90%. Figure A111 presents in a single chart the three distributions for changes in GHG emissions obtained from canola, tallow, and waste oil biodiesel compared to LSD.

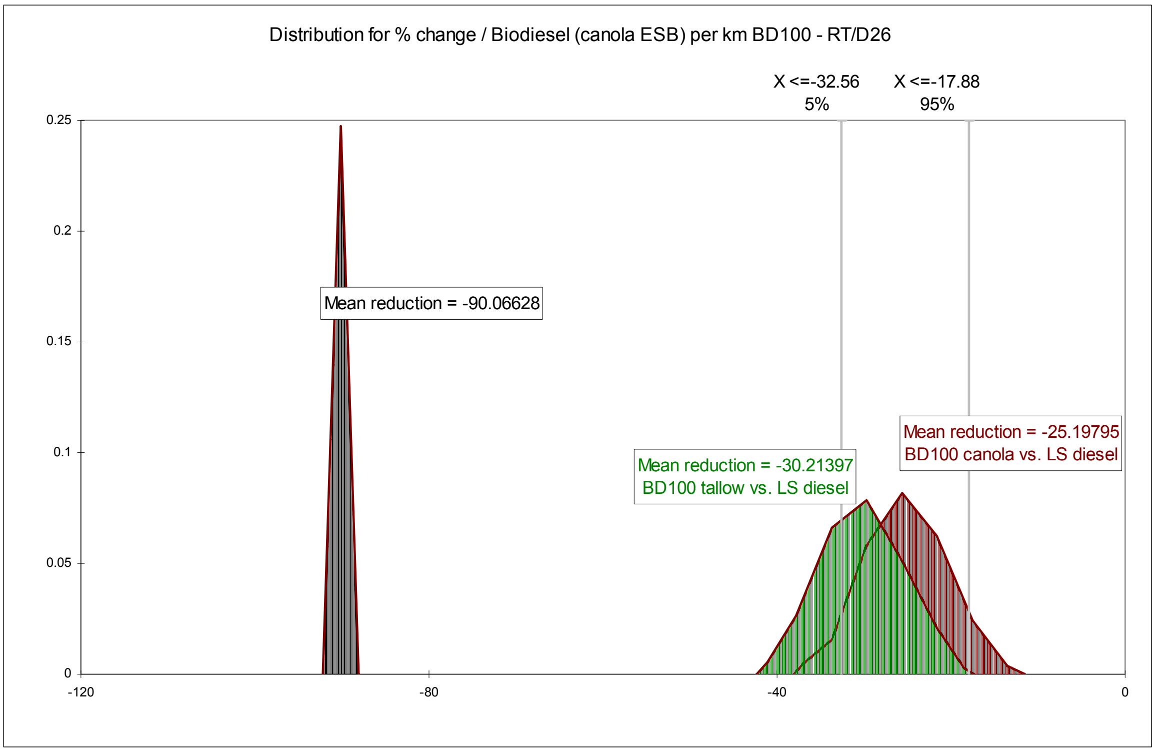


Figure A111. Distribution of percentage change in GHG emissions per km from BD100 compared to LSD (rigid truck)

Air pollutant emissions

The distributions of changes in CO emissions from use of biodiesel B100 are provided in Figures 112 to 114.

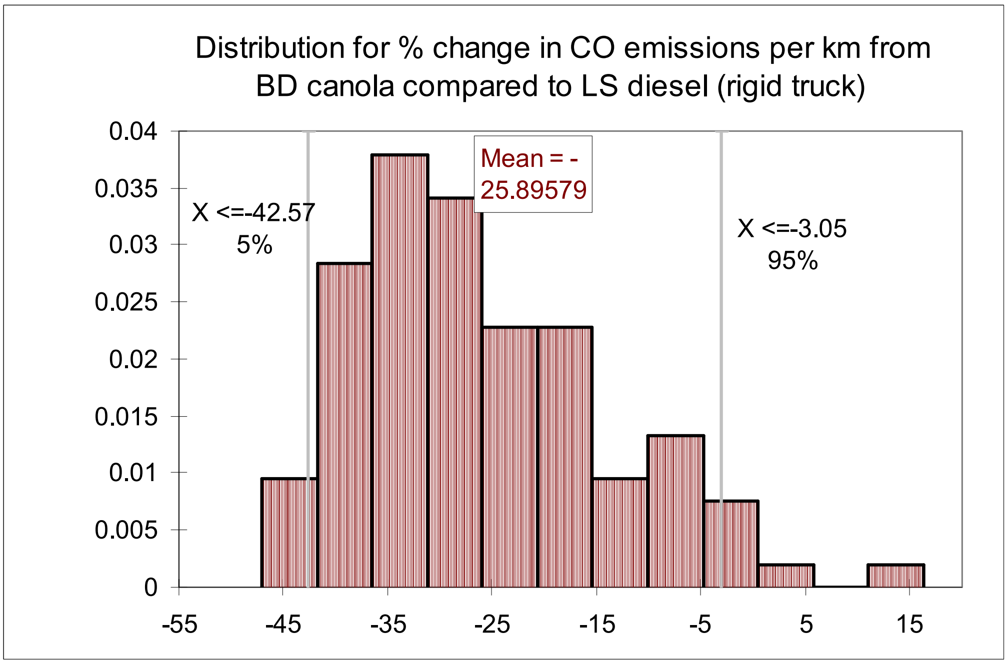


Figure A112. Distribution of percentage change in CO emissions per km from BD100 (canola) compared to LSD (rigid truck)

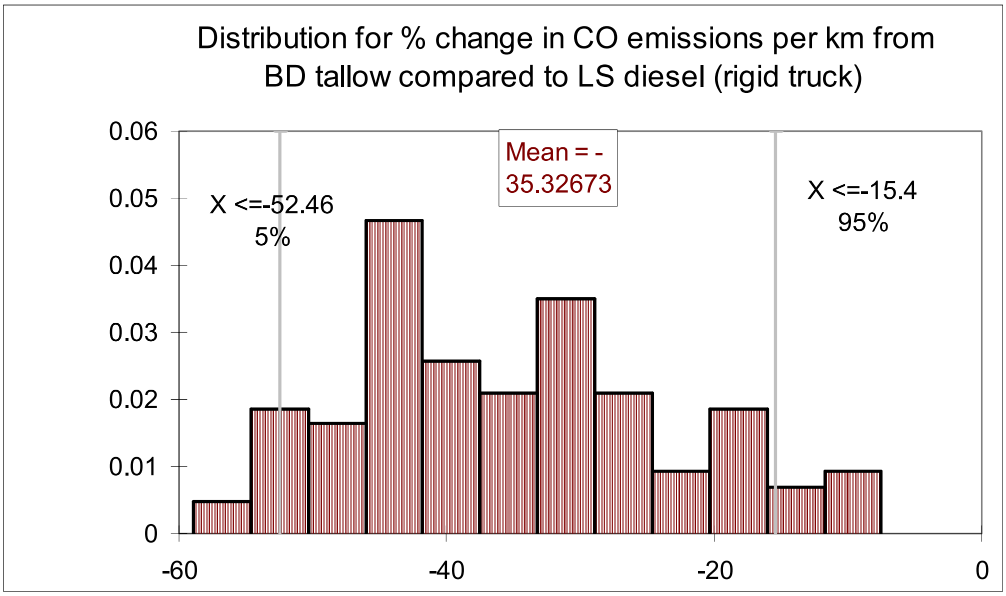


Figure A113. Distribution of percentage change in CO emissions per km from BD100 (tallow) compared to LSD (rigid truck)

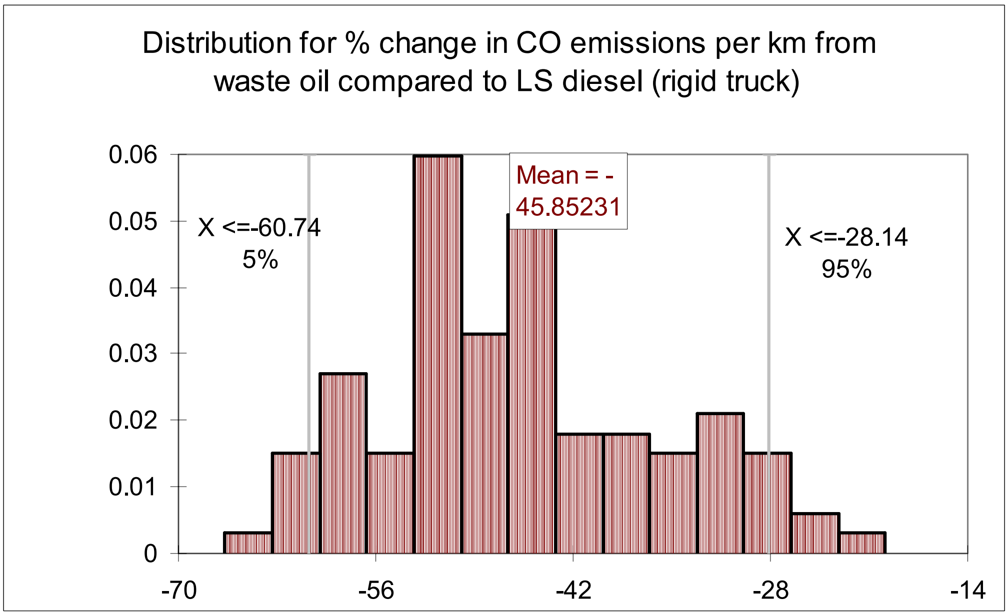


Figure A114. Distribution of percentage change in CO emissions per km from BD100 (waste oil) compared to LSD (rigid truck)

The highest savings result from the use of waste oil biodiesel: the range of savings is between 28.1% and 60.7%, with an average of 45.9% (Figure A114).

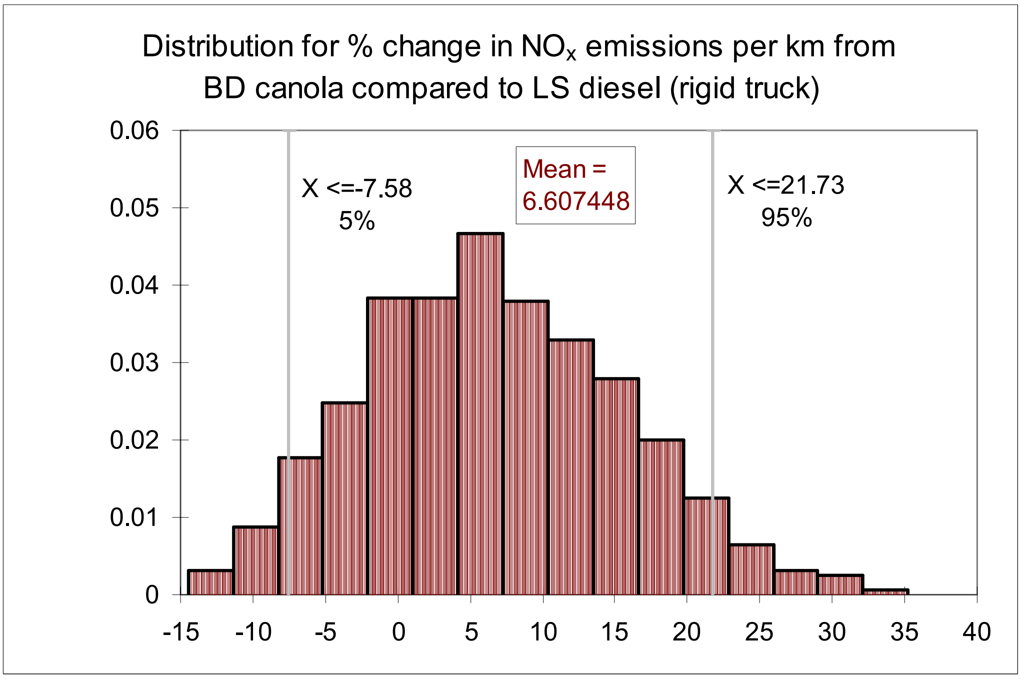


Figure A115. Distribution of percentage change in NO_x emissions per km from BD100 (canola) compared to LSD (rigid truck)

Figure A115 indicates a 24.3% probability of obtaining reductions in NO_x when using BD100 canola.

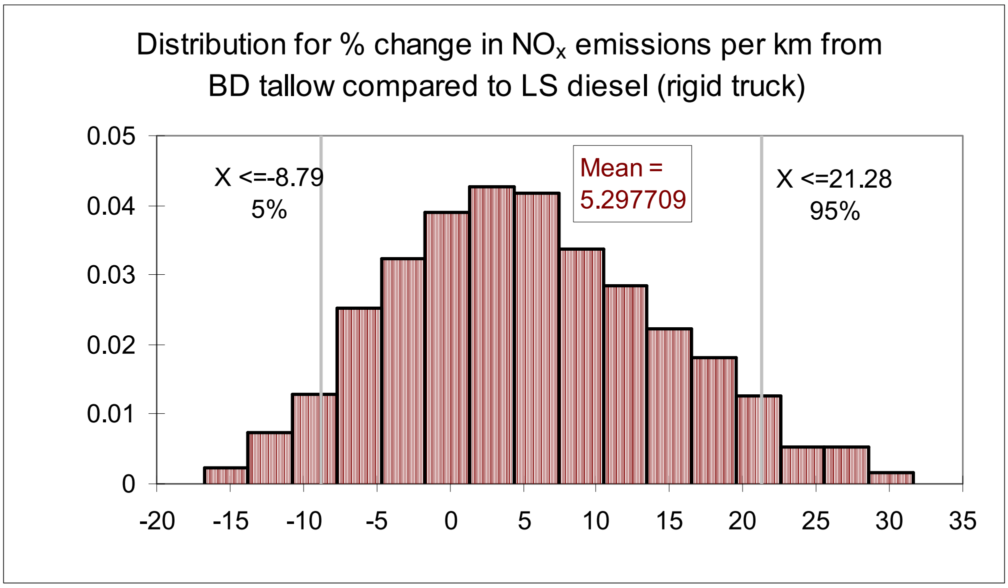


Figure A116. Distribution of percentage change in NO_x emissions per km from BD100 (tallow) compared to LSD (rigid truck)

Figure A116 shows an average increase in NO_x emissions from BD100 tallow of 5.3% and a probability of 30.9% of obtaining reductions in NO_x when using BD100 tallow.

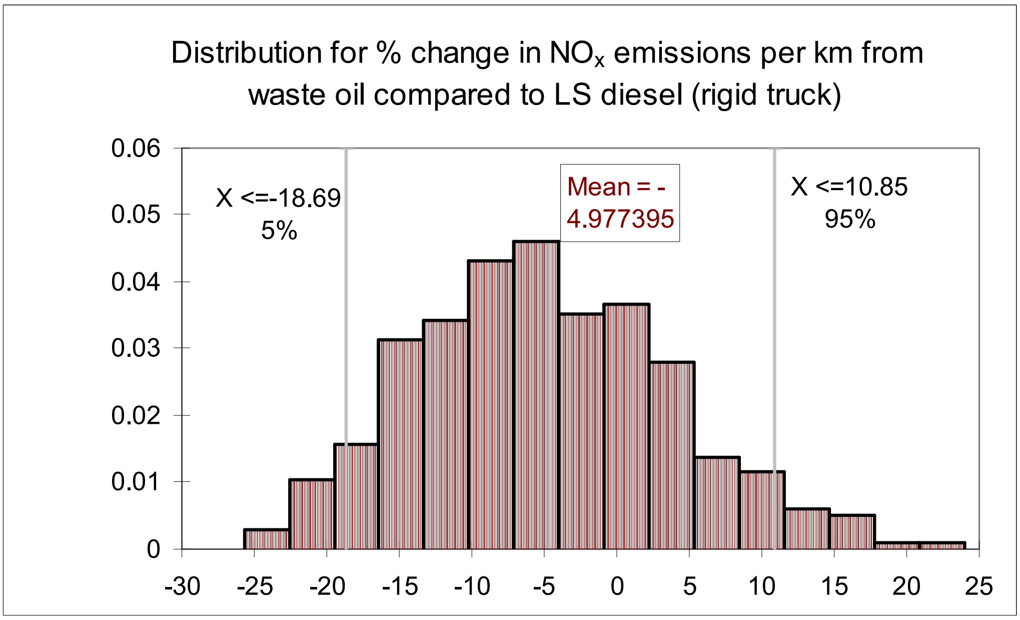


Figure A117. Distribution of percentage change in NO_x emissions per km from BD100 (waste oil) compared to LSD (rigid truck)

Figure A117 indicates that the use of BD100 waste oil would lead to average savings in NO_x emissions per km of 4.98%, but there is still a 28.2% probability of obtaining increases in NO_x emissions when using BD100 waste oil.

Figures 118 to 120 present the reductions in NMVOC emissions per km resulting from use of BD100.

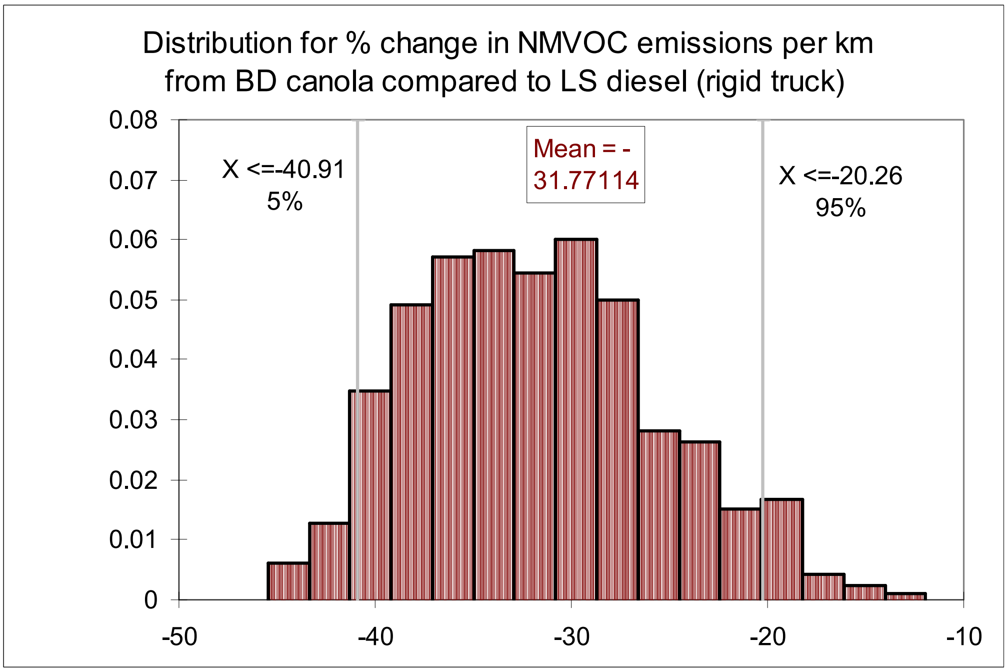


Figure A118. Distribution of percentage change in NMVOC emissions per km from BD100 (canola) compared to LSD (rigid truck)

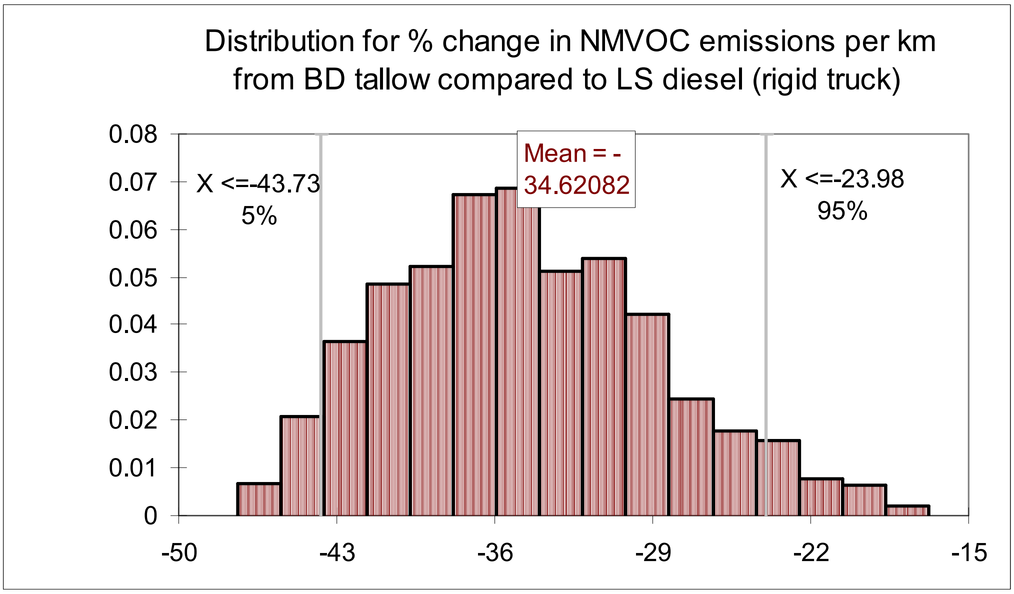


Figure A119. Distribution of percentage change in NMVOC emissions per km from BD100 (tallow) compared to LSD (rigid truck)

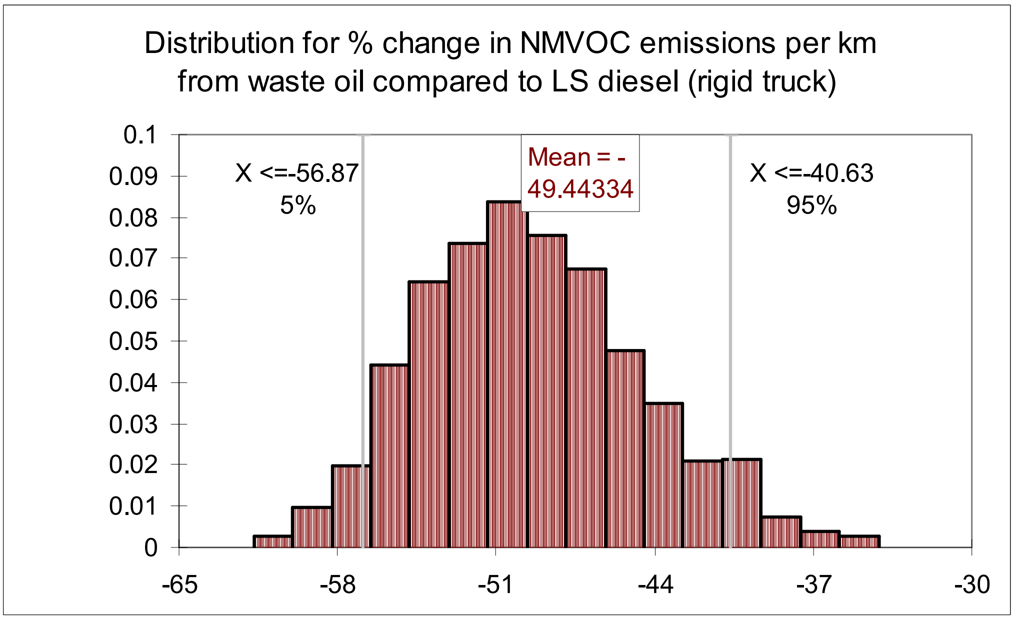


Figure A120. Distribution of percentage change in NMVOC emissions per km from BD100 (waste oil) compared to LSD (rigid truck)

Figure A121 to Figure A123 show the distributions of reductions in PM emissions per km resulting from the use of canola, tallow, and waste oil biodiesel, compared with LSD.

The charts illustrate that the reductions are statistically significant at the 0.05 level.

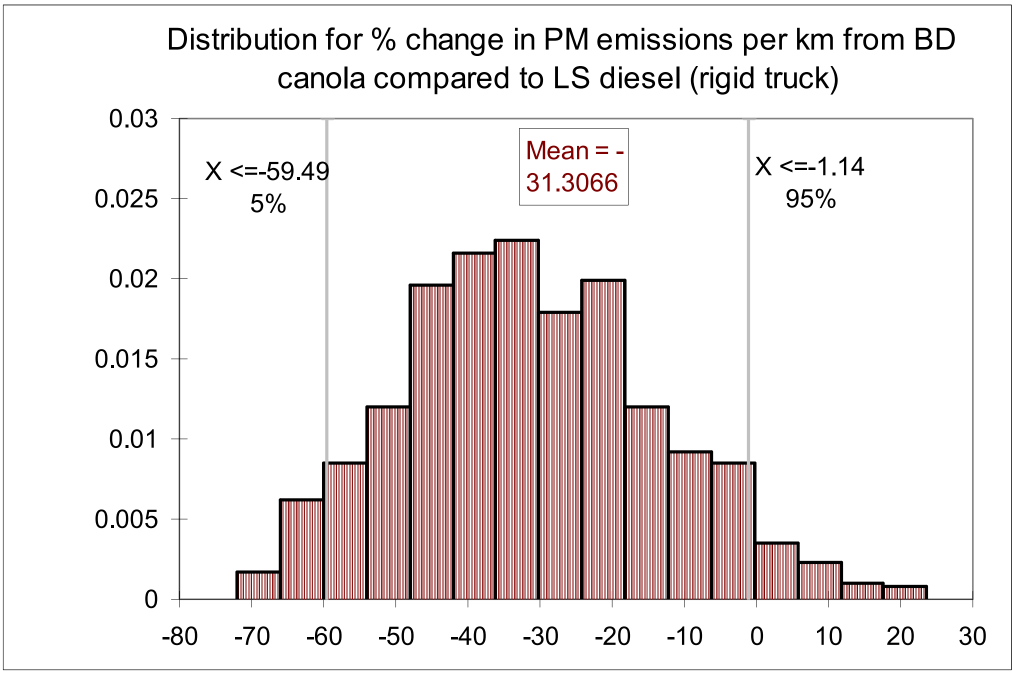


Figure A121. Distribution of percentage change in PM emissions per km from BD100 (canola) compared to LSD (rigid truck)

Figure A121 indicates only a probability of 4.4% of obtaining an increase in PM emissions with BD100 canola, when compared with LSD.

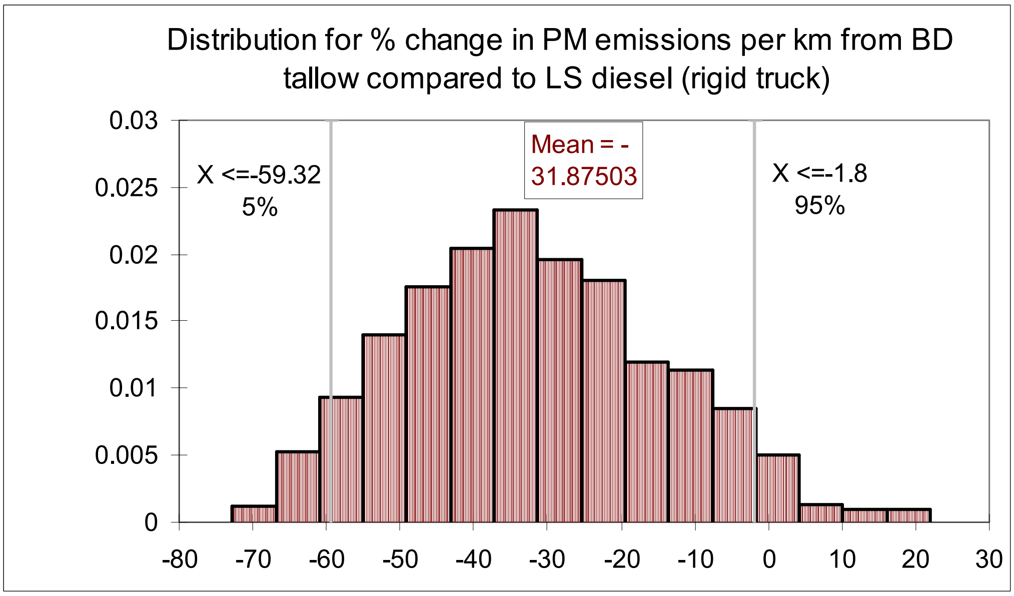


Figure A122. Distribution of percentage change in PM emissions per km from BD100 (tallow) compared to LSD (rigid truck)

Figure A122 shows that there is only a 4.2% probability of obtaining an increase in PM emissions with BD100 tallow, and Figure A123 indicates a probability of 1.4% for an increase in PM emissions with BD100 waste oil.

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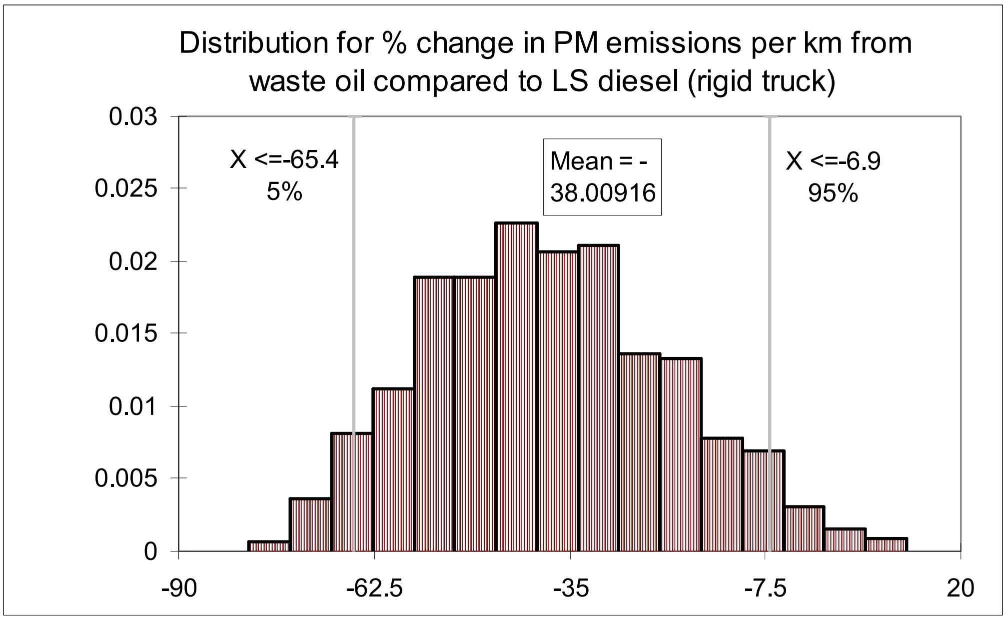

Figure A123. Distribution of percentage change in PM emissions per km from BD100 (waste oil) compared to LSD (rigid truck)

Upon request CSIRO can provide the distributions for changes in emissions per km for all biodiesel blends compared to the three base diesel fuels: LSD, ULS, and XLS.

Uncertainty in environmental costs

Using Watkiss (2002) to value pollutants (Tables 64 and 65 of the main report) and the uncertainties of emissions presented above, we determined the total emission costs per km of all fuels as stochastic variables.

Figures 124 to 128 provide a sample of the distributions of pollution costs, calculated on a per km basis for pure biodiesel and LSD.

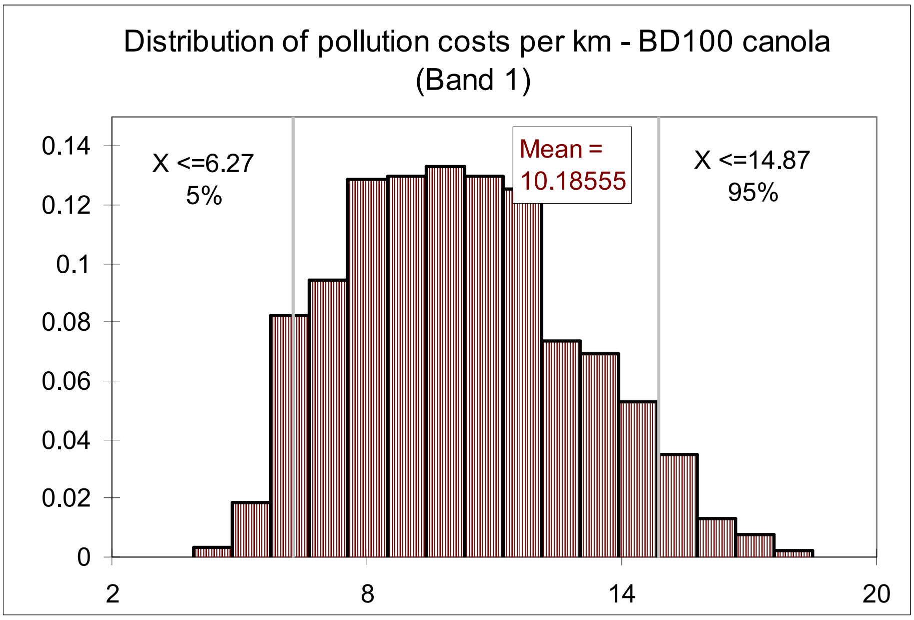

Figure A124. Distribution of pollution cost per km from BD100 (canola) – Band 1

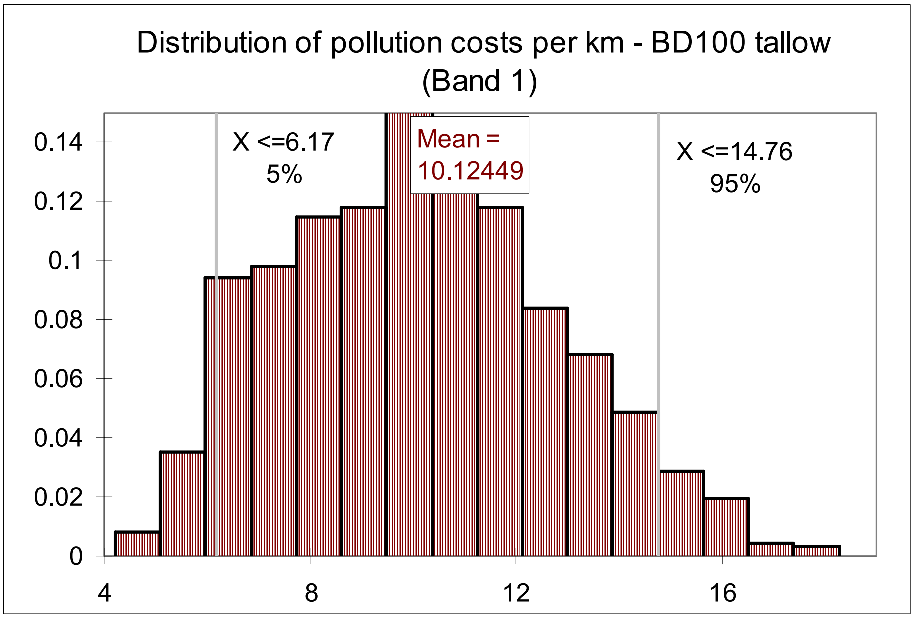

Figure A125. Distribution of pollution cost per km from BD100 (tallow) – Band 1

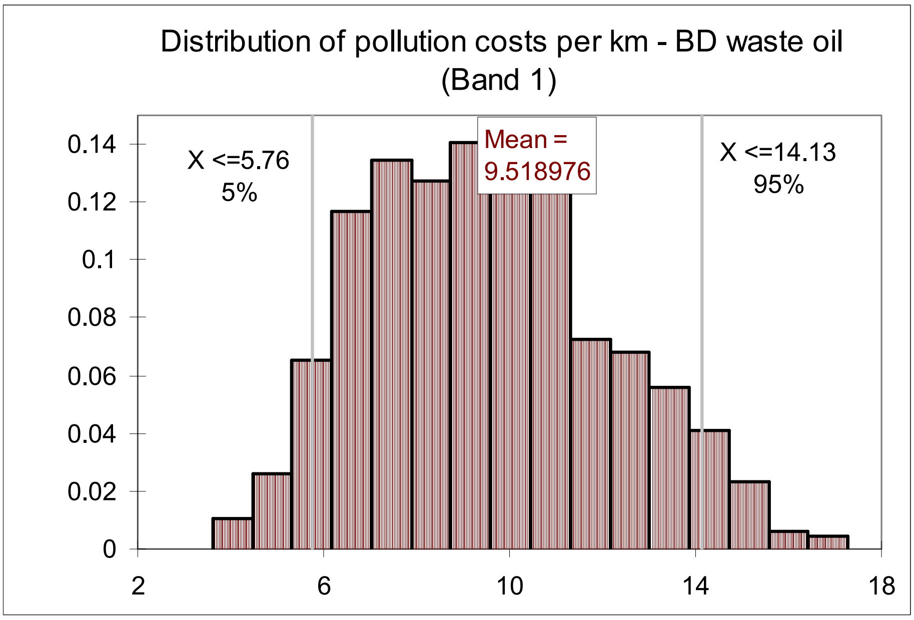

Figure A126. Distribution of pollution cost per km from BD100 (waste oil) – Band 1

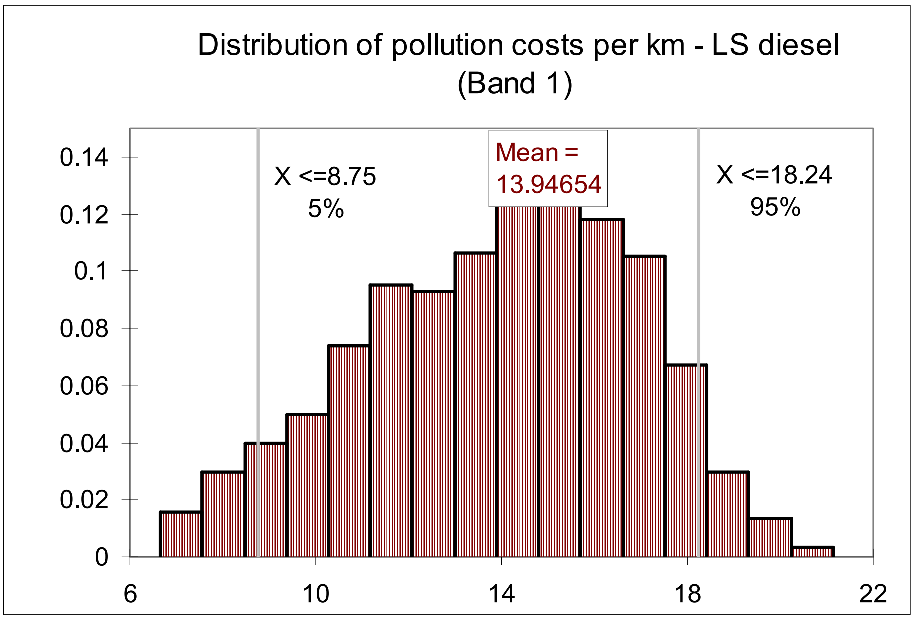

Figure A127. Distribution of pollution cost per km from LS Diesel – Band 1

In Figure 128, for comparison purposes, we present the distribution of health costs for pure biodiesel (BD100) and LSD in Band 4. As expected, the lower unit health costs in band 4 lead to lower health costs per km, and the use of pure biodiesel leads to lower health costs than LSD.

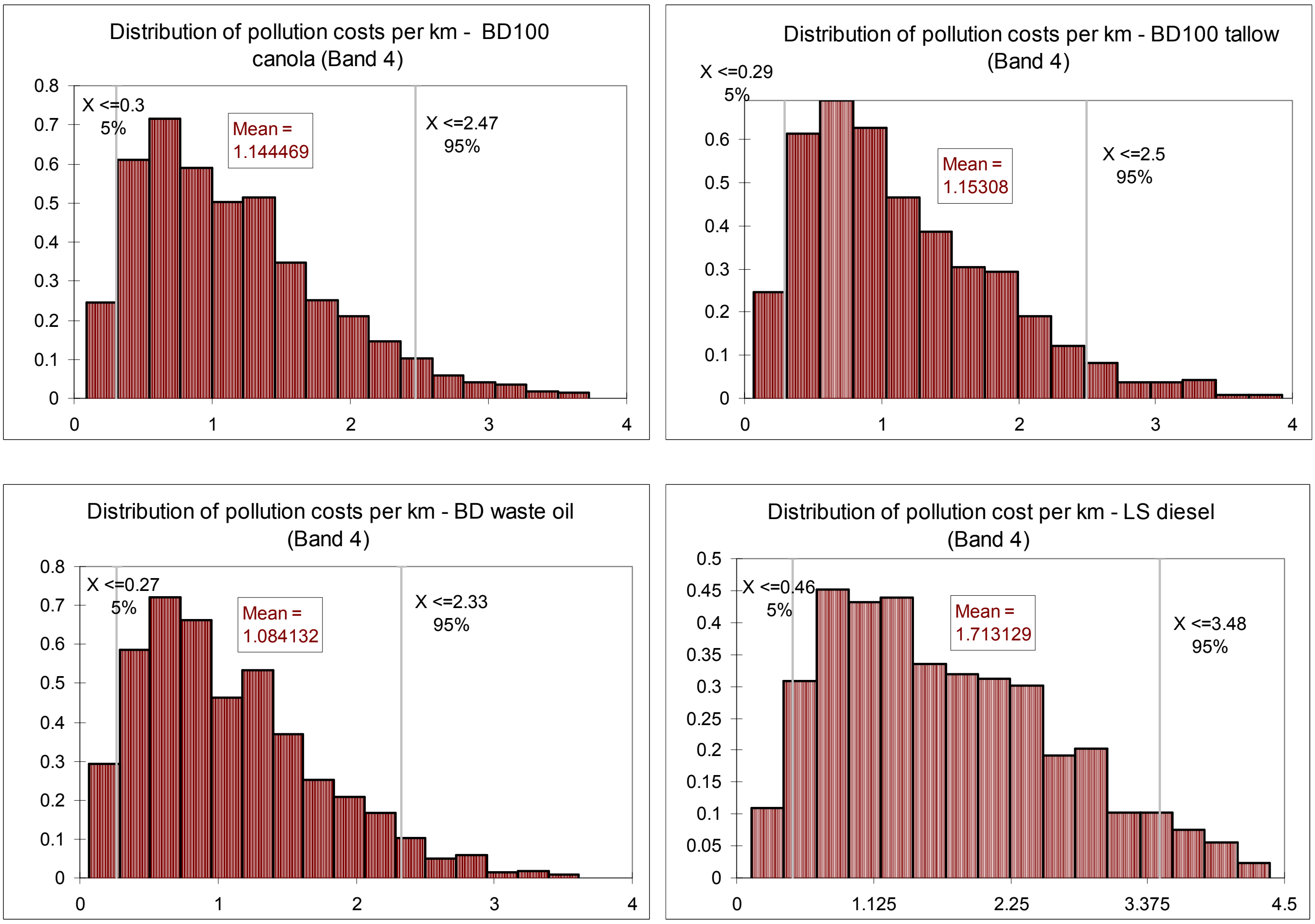

Figure A128. Distribution of pollution cost per km from BD100 and LSD – Band 4

The distributions of health costs (based on average values from Watkiss, 2002) and the distributions of air pollutant emissions per km were use to estimate the future health costs in the two situations: base/reference case of fuel market share and the introduction of 350ML biofuels program.

The forecast has been generated to the year 2020 using deterministic traffic data provided by BTRE.

Table 73 of the report presents the total and average health costs per L in 2010. The distributions for total health costs in 2010 are provided in the Figures 21 to 23 of the main report (Section 13.2.2).

CSIRO can make available all distribution of health costs per km for ethanol, unleaded petrol, 20% and 5% biodiesel blends with ULS and XLS base diesel fuel and the distribution of health costs for entire period of analysis.