Harmonisation and extension of the bioenergy inventories and assessment

End report

EMPA

Materials Sci ence & Technolog y

Schweizerische Eidgenossenschaft
Confédération suisse
Confederazione Svizzera
Confederaziun svizra

Eidgenössisches
Volkswirtschaftsdepartement EVD
Forschungsanstalt
Agroscope Reckenholz-Tänikon ART

PAUL SCHERRER INSTITUT

PSI

Doka Ökobilanzen

Autoren:
Mireille Faist Emmenegger, Empa, mireille.faist@empa.ch
Simon Gmünder, Empa, simon.gmuender@empa.ch
Jürgen Reinhard, Empa, juergen.reinhard@empa.ch
Rainer Zah, Empa, rainer.zah@empa.ch
Thomas Nemecek, ART, thomas.nemecek@art.admin.ch
Julian Schnetzer, ART, julian.schnetzer@art.admin.ch
Christian Bauer, PSI, christian.bauer@psi.ch
Andrew Simons, PSI andrew.simons@psi.ch
Gabor Doka, Doka Ökobilanzen, gabor@doka.ch

Abstract

This study extended and updated the ecoinvent inventories dating back from 2007. Various new crop inventories, conversion processes for biofuel production and unconventional fossil pathways have been introduced, the nitrogen fluxes and the land use modelling have been updated, and state-of-the-art impact assessment methods have been used for calculating the LCA. The have results have been compared with those from the 2007 study of Zah et al. (2007) and uncertainty has been thoroughly analysed.

The results of the selected ILCD-Midpoint indicators distinctly show that for many of the indicators, the biofuel value chains have higher impact values than the fossil reference, except for the methane pathways. Only climate change and ozone depletion are more favourable in most of the biofuel pathways than the fossil reference. Higher values for biofuel production pathways are found in indicators relevant for agricultural processes (eutrophication, acidification, water depletion, ecotoxicity, land use).

Although biofuels can lead to a reduction of fossil fuel use and of greenhouse gas emissions, they often shift environmental burdens towards land use-related impacts. Indeed, only very few biofuel pathways show lower or at least no higher impacts for all indicators than the fossil fuels. The most promising pathways are those based on methanisation of residues or on reforestation activities.

Generally, the study confirms the high diversity in the impact patterns of biofuel pathways and therefore the necessity of assessing biofuel projects with specific data. However, uncertainty of the results is high due to lack of data and modelling uncertainties. There is for example a need for more specific modelling of agricultural N_2O. This uncertainty should lead to general caution when promoting biofuels.

Executive summary

Einleitung

Die Lebenszyklusanalyse (LCA) hat sich in den letzten Jahren in der Schweiz und international als wichtiges Werkzeug für die Beurteilung von Bioenergie etabliert und wurde in mehreren Zertifizierungsverfahren eingeführt. Die Verfügbarkeit von öffentlich zugänglichen, verlässlichen und überprüfbaren Daten ist eine Grundvoraussetzung für die Unterstützung solcher Beurteilungen.

Der Bioenergiemarkt ist sehr dynamisch und wächst in Europa aufgrund der Beimischziele der Renewable Energy Directive (RED) stark. Die ökologischen Auswirkungen der Produktion und Nutzung von Energie aus Biomasse sind einerseits stark von neuen Energiepflanzen, landwirtschaftlichen Verfahren und Umwandlungstechnologien beeinflusst. Andererseits werden die Methoden für die Messung von Materialflüssen und Emissionen und für die Bewertung ihrer Auswirkungen auf die Umwelt ständig weiterentwickelt. Folglich ist es für die Beurteilung von Bioenergie von zentraler Bedeutung, sich auf den aktuellsten Stand der Technik, sowohl von den Daten als auch von der Methodik her, stützen zu können. Ziel dieses Projekt ist deshalb die Verbesserung, Harmonisierung und Erweiterung der ecoinvent-Inventare zur Bioenergie.

Methodik

In diesem Projekt wurden verschiedene Aktualisierungen der Bioenergie-Inventare parallel durchgeführt. Einerseits wurde die Berechnung von Stickstoffemissionen in der Landwirtschaft und von Treibhausgas-Emissionen aus Landnutzungsänderung harmonisiert. Andererseits wurde die bestehende Parametrisierung der Kehrichtverbrennung weiter entwickelt. Zusätzlich wurden weitere neue Inventare für Energiepflanzen und Umwandlungsprozesse modelliert.

Für Stickstoffemissionen in der Landwirtschaft (Nitrat, Ammoniak, Lachgas) wurden einheitliche Berechnungsmodelle festgelegt. So wird für Nitrat in der Schweiz und Europa das SALCA-Modell angewendet; für den Rest der Welt gilt einheitlich das SQCB-Modell. Die Stickstoffemissionen wurden entsprechend für alle existierenden ecoinvent-Pflanzeninventare neu berechnet.

Eine einheitliche Vorgehensweise wurde auch für die Berechnung der Emissionen aus der Landnutzungsänderung definiert und angewandt. Hier gelten die Modelle und die Werte von IPCC. Neu werden alle Biomasse-Kompartimente (above ground biomass, below ground biomass, dead organic matter) berücksichtigt und einheitlich auf 20 Jahre abgeschrieben; dabei wird zwischen Kohlenstoffabbau und -speicherung differenziert.

Neue Inventare für Energiepflanzen (Jatropha, Salix, Miscanthus, Alfalfa) sowie für Umwandlungsprozesse (Wärme-Kraft-Kopplung, Gas- und Dampfkraftwerke) wurden entwickelt. Zusätzlich wurde die Produktion von Benzin und Diesel aus kanadischen Ölsänden untersucht. Ziel war es, die Sensitivität der Umweltauswirkungen gegenüber unterschiedlichen fossilen Treibstoffen zu bestimmen.

Das ecoinvent-Modell der KVA-Verbrennung wurde in den Bereichen Dioxinemissionen, Energieausbeuten und installierte DeNOx-Technologie aktualisiert. Das Modell wurde bei der Berechnung des ehemals konstanten, internen Energieverbrauchs sowie der abfallspezifischen Allokation von ehemals konstanten prozessspezifischen Emissionen verfeinert. Für den in dieser Studie betrachteten Biomasseabfall wurde der Wassergehalt und somit der Heizwert aufgrund einer Literaturstudie untersucht und in drei Varianten berechnet: Gartenabfälle (eher trocken), Küchenabfälle (eher feucht), sowie durchschnittliche Biomasseabfälle.

Als funktionelle Einheit für die Berechnung der LCA von Biotreibstoffen wurde ein in einem durchschnittlichen Personenwagen gefahrener Kilometer gewählt. Mehrere Bewertungsmethoden wurden sowohl auf Ebene der Charakterisierung als auch auf Ebene der Einzelergebnisse angewendet. Die Berechnung mit den gleichen Methoden wie in der Empa Biotreibstoffstudie von 2007 (Zah, R. *et al.* 2007) erlaubte es, Trends in den

Inventardaten zu analysieren. Zusätzlich wurden die Berechnungen mit der aktuellsten Version der von einem holländischen Forscherteam entwickelten Bewertungsmethode ReCiPe 2008 durchgeführt, um Unterschiede aufgrund der Bewertungsmethodik zu eruieren.

Für sämtliche Produktionsketten wurden die Unsicherheiten mittels Monte Carlo Analysen berechnet. Die Informationen dazu stammen aus der ecoinvent-Datenbank. Alle Berechnungen wurden mit Simapro 7.3.3 durchgeführt.

Grenzen der Arbeit

Die Modellierung mit ecoinvent Version 3 verwendet neue Systemmodelle und wird sich deshalb von den früheren Versionen unterscheiden. Aufgrund der Verzögerung dieser wichtigen Aktualisierung der Datenbank war es entgegen der ursprünglichen Planung nicht möglich, ecoinvent Version 3 (v3) für die Berechnungen in diesem Projekt zu benutzen. Stattdessen wurde Simapro noch mit der ecoinvent v2.2 angewendet. Entsprechend können sich die Resultate gegenüber der finalen Berechnung mit Version 3 unterscheiden.

Aufgrund der begrenzten Ressourcen war es weiter nicht möglich, alle Flüsse in den landwirtschaftlichen Inventaren zu aktualisieren (z.B. eine Harmonisierung der Phosphatemissionen in einem ähnlichen Verfahren wie für Stickstoff). Die neuen Transportinventare für EURO4/EURO5 sind noch in der Entwicklungsphase und konnten hier nicht benutzt werden. Trotz dieser Einschränkungen stellen die Ergebnisse aber einen wesentlichen Schritt in der Verbesserung der Bioenergie-Datenbank dar.

Wie umweltfreundlich sind Biotreibstoffe?

Biotreibstoff-Produktionsketten zeichnen sich durch eine hohe Variabilität aus. Nicht nur der Rohstoff, sondern auch Herkunft, Anbaumethoden und Umwandlungsprozesse beeinflussen die Resultate stark. Auch wenn gewisse Tendenzen erkennbar sind, spiegeln die Resultate doch primär die Eigenschaften der jeweiligen individuellen Kette. Dies zeigt sich in Tabelle 1, in der die Biotreibstoffketten bezüglich ihren Umweltauswirkungen (Resultate der Midpoint-Indikatoren) mit verschiedenen fossilen Varianten verglichen werden. Die Unterschiede bei gleichem Rohstoff und gleichem Treibstoff werden zum Beispiel bei den Resultaten für Treibhausgaspotential bei den Soja-Methylester-Ketten oder bei den sehr unterschiedlichen Resultaten für den Wasserverbrauch bei Jatropha ersichtlich.

Eine Hauptmotivation für die Produktion von Biotreibstoffen ist es, fossile Treibstoffe zu substituieren und dadurch fossile CO_2-Emissionen mit solchen aus biogenen Quellen sowie fossile Energieressourcen durch erneuerbare zu ersetzen. Allerdings führt die Biotreibstoffproduktion nicht immer zur Reduktion von Treibhausgasemissionen im Vergleich zum fossilen System, weil die Erschliessung neuer Landflächen für deren Anbau oder der Düngemitteleinsatz in der Landwirtschaft hohe Treibhausgasemissionen verursachen können.

Für Umweltwirkungen wie Eutrophierung, Versauerung oder Ökotoxizität, die stark von der Landwirtschaft geprägt sind, liegen die Werte für Biotreibstoffe meistens höher als für die fossile Referenz, mit der Ausnahme von Biogas aus Bioabfällen.

Im Allgemeinen zeigt sich eine Verlagerung der Umweltbelastungen, wenn fossile Treibstoffe durch Biotreibstoffe ersetzt werden. Während letztere in der Regel weniger fossile Energie brauchen und weniger Treibhausgasemissionen verursachen als die fossile Referenz, sind ihre Umweltauswirkungen für die meisten anderen Indikatoren deutlich höher, insbesondere für jene, die mit der landwirtschaftlichen Produktion verbunden sind wie Eutrophierung oder Toxizität.

Tabelle 1: **Resultate pro Fahrzeugkilometer der Biotreibstoff- und fossile Treibstoffketten für einige ausgewählte Indikatoren (für eine vollständigere Darstellung s. Kap. 4.3.1).**

	Treibhaus-gaspoten-tial 100a	Süsswasser Eutrophie-rung	Meer-wasser Eutro-phierung	Terres-trische Ver-sauerung	Öko-toxizität	Mensch-liche Toxizität, Krebs	Mensch-liche Toxizität, Nicht-Krebs	Wasser-ressourcen Verbrauch	Fossiler, mineral. und bioge-ner Res-sourcen-abbau
	kg CO2 eq	kg P eq	kg N eq	molc H+ eq	CTUe	CTUh	CTUh	m3 water eq	kg Sb eq
Raps ME IP/CH	2.1E-01	7.7E-05	3.1E-03	3.2E-03	7.7E-01	2.2E-08	5.7E-07	1.1E-04	7.0E-06
Raps ME EXT/CH	2.1E-01	6.4E-05	3.6E-03	4.5E-03	7.1E-01	2.0E-08	6.6E-07	1.1E-04	7.0E-06
Raps ME conv/DE	1.7E-01	7.5E-05	7.2E-04	1.3E-03	4.1E-01	2.2E-08	2.3E-08	1.1E-04	6.9E-06
Raps ME conv/FR	2.1E-01	1.0E-04	2.2E-03	2.9E-03	1.1E+00	2.4E-08	6.7E-08	1.1E-04	7.0E-06
Soja ME BR	8.4E-01	6.6E-05	1.8E-03	1.3E-03	5.3E+00	2.2E-08	-1.8E-07	9.7E-05	8.8E-05
Soja ME US	1.4E-01	6.3E-05	1.6E-03	1.0E-03	2.3E-01	2.0E-08	-1.7E-07	9.6E-05	6.5E-06
Jatropha ME EXT/IN	-2.2E-01	6.2E-05	2.6E-03	3.6E-03	6.3E-01	2.2E-08	5.4E-07	6.1E-04	5.6E-05
Jatropha ME INT/IN	1.4E-01	1.0E-04	1.8E-03	1.1E-02	9.5E-01	4.5E-08	6.5E-07	3.3E-03	1.7E-05
Jatropha ME Zaun/AFR	-1.4E-02	4.1E-05	4.2E-04	7.5E-04	2.8E-01	1.8E-08	1.8E-08	8.7E-05	6.2E-06
Palmöl ME MY	3.3E-01	5.2E-05	1.5E-03	1.4E-03	4.7E-01	2.0E-08	3.4E-08	3.2E-03	3.5E-05
Palmöl ME CO	8.4E-02	4.7E-05	3.7E-03	1.2E-03	5.3E-01	1.9E-08	7.7E-08	3.4E-04	9.9E-06
Zuckerrohr-Molasse ETOH BR	1.2E-01	5.5E-05	9.3E-04	2.2E-03	8.6E-01	2.4E-08	3.6E-07	3.6E-04	6.8E-06
Zuckerrohr ETOH BR	1.2E-01	5.5E-05	8.1E-04	2.1E-03	7.5E-01	2.3E-08	2.7E-07	5.0E-04	6.9E-06
Zuckerrohr-Molasse ETOH CO	1.3E-01	4.5E-05	7.3E-04	2.2E-03	7.1E-01	2.1E-08	1.8E-07	1.6E-02	6.6E-06
Mais ETOH US	2.7E-01	9.8E-05	1.5E-03	2.5E-03	1.3E+00	2.3E-08	-3.7E-08	4.5E-04	7.0E-06
Roggen ETOH CONV/RER	3.1E-01	1.2E-04	5.9E-03	2.9E-03	1.5E+00	2.9E-08	6.5E-07	2.2E-04	7.8E-06
Zuckerrüben ETOH IP/CH	1.2E-01	4.6E-05	5.7E-04	1.2E-03	2.5E-01	1.9E-08	-8.4E-08	1.2E-04	6.5E-06
Zuckerhirse ETOH CN	1.3E-01	6.9E-05	4.3E-04	1.9E-03	7.6E-01	2.2E-08	1.1E-07	5.2E-03	6.8E-06
Weizen ETOH US	3.4E-01	2.7E-04	1.4E-03	4.0E-03	9.4E-02	-1.1E-08	-3.7E-07	4.1E-03	8.4E-06
Methan 96%, Alfalfa	2.3E-01	7.3E-05	2.1E-04	1.6E-03	2.6E-01	2.0E-08	-6.3E-08	2.1E-04	6.7E-06
Methan 99%, Klärschlamm	1.4E-01	4.6E-05	8.9E-05	5.3E-04	3.0E-01	1.9E-08	1.7E-08	1.4E-04	6.2E-06
Methan 96%, Holzschnitzel	7.0E-02	3.9E-05	8.6E-05	5.2E-04	2.8E-01	1.8E-08	1.8E-08	9.1E-05	6.4E-06
Methane 96%, Gülle	9.2E-02	4.9E-05	1.4E-04	6.8E-04	3.6E-01	2.1E-08	9.2E-08	1.3E-04	7.4E-06
Erdgas	2.6E-01	3.7E-05	9.4E-05	5.7E-04	2.8E-01	1.8E-08	1.7E-08	8.7E-05	6.0E-06
Fossiles Benzin, schwefelarm	3.2E-01	4.0E-05	1.6E-04	8.0E-04	2.9E-01	1.8E-08	2.0E-08	9.2E-05	6.1E-06
Fossiles Diesel, schwefelarm	2.7E-01	3.9E-05	2.1E-04	8.2E-04	2.9E-01	1.8E-08	2.0E-08	8.8E-05	6.1E-06
Diesel, SCO	3.0E-01	4.0E-05	2.0E-04	1.3E-03	3.4E-01	2.1E-08	2.1E-08	3.9E-02	6.0E-06
Benzin, SCO	3.5E-01	4.2E-05	1.5E-04	1.4E-03	3.5E-01	2.2E-08	2.0E-08	4.3E-02	6.0E-06

Farblegende:

	Höher als 95% der Referenz		5%-40% tiefer als Referenz		5%-67% höher als Referenz
	Referenz		Mehr als 40% tiefer als Referenz		67% höher als Referenz
	Kleiner als 105% der Referenz				

Wie verteilen sich die Umweltbelastungen entlang der Wertschöpfungskette?

Bei Biotreibstoffen aus landwirtschaftlichen Rohstoffen dominiert der Anbau die Umweltbelastungen (Abbildung 1, grün). Für die Treibhausgasemissionen sind dafür primär das bei der Landumwandlung freigesetzte CO_2 und die Lachgasemissionen verantwortlich. Die Nährstoff- bzw. Pestizidemissionen spielen dagegen für Eutrophierung, Versauerung bzw. Ökotoxizität eine wichtige Rolle. An zweiter Stelle folgen die Umweltauswirkungen der Infrastruktur (Abbildung 1, grau) oder bei der Biogasproduktion der Methanschlupf im

Aufbereitungsprozess (Abbildung 1, gelb). Die Muster sind dabei bei beiden Endpoint-Methoden, UBP und ReCiPe, sehr ähnlich.
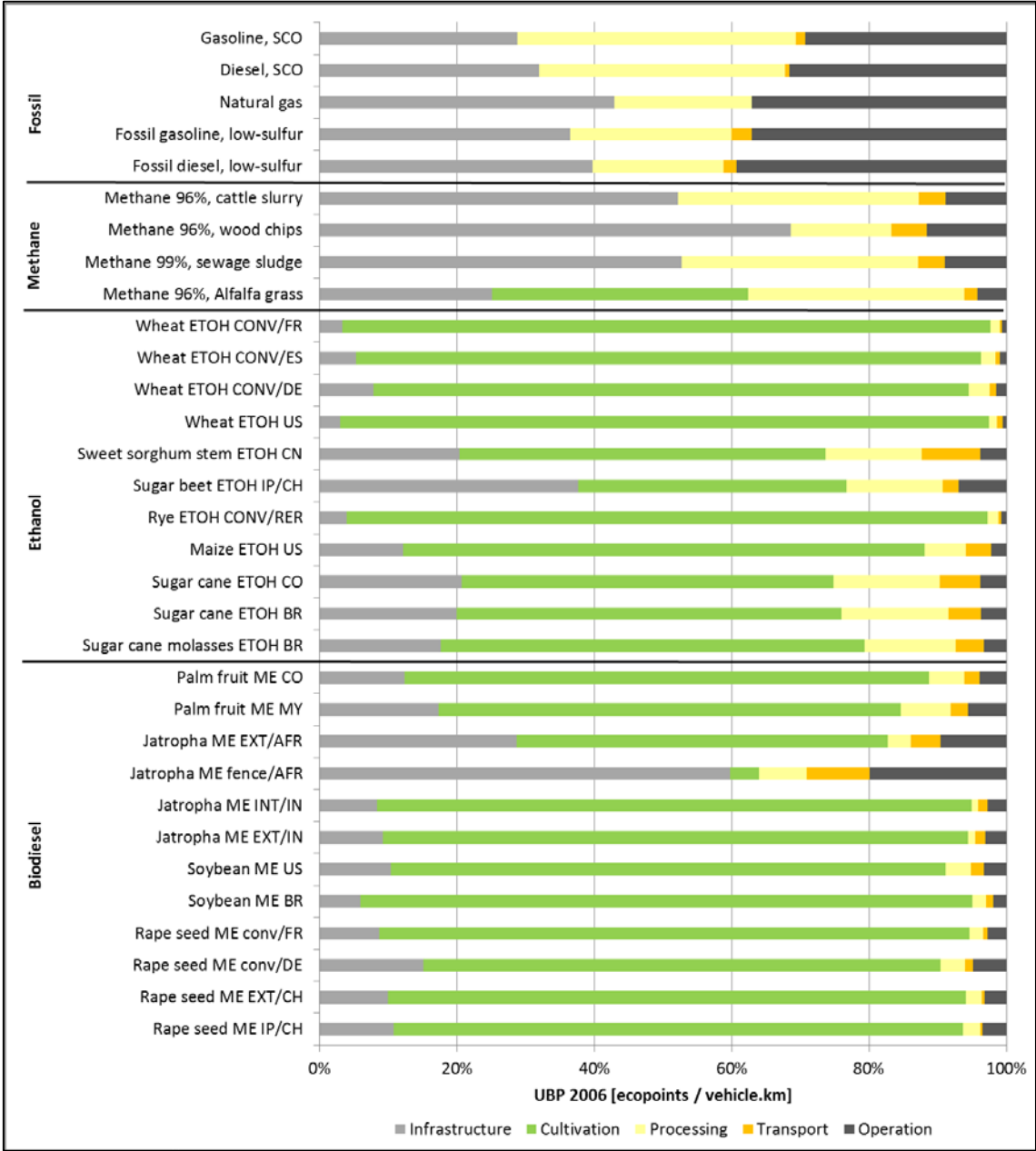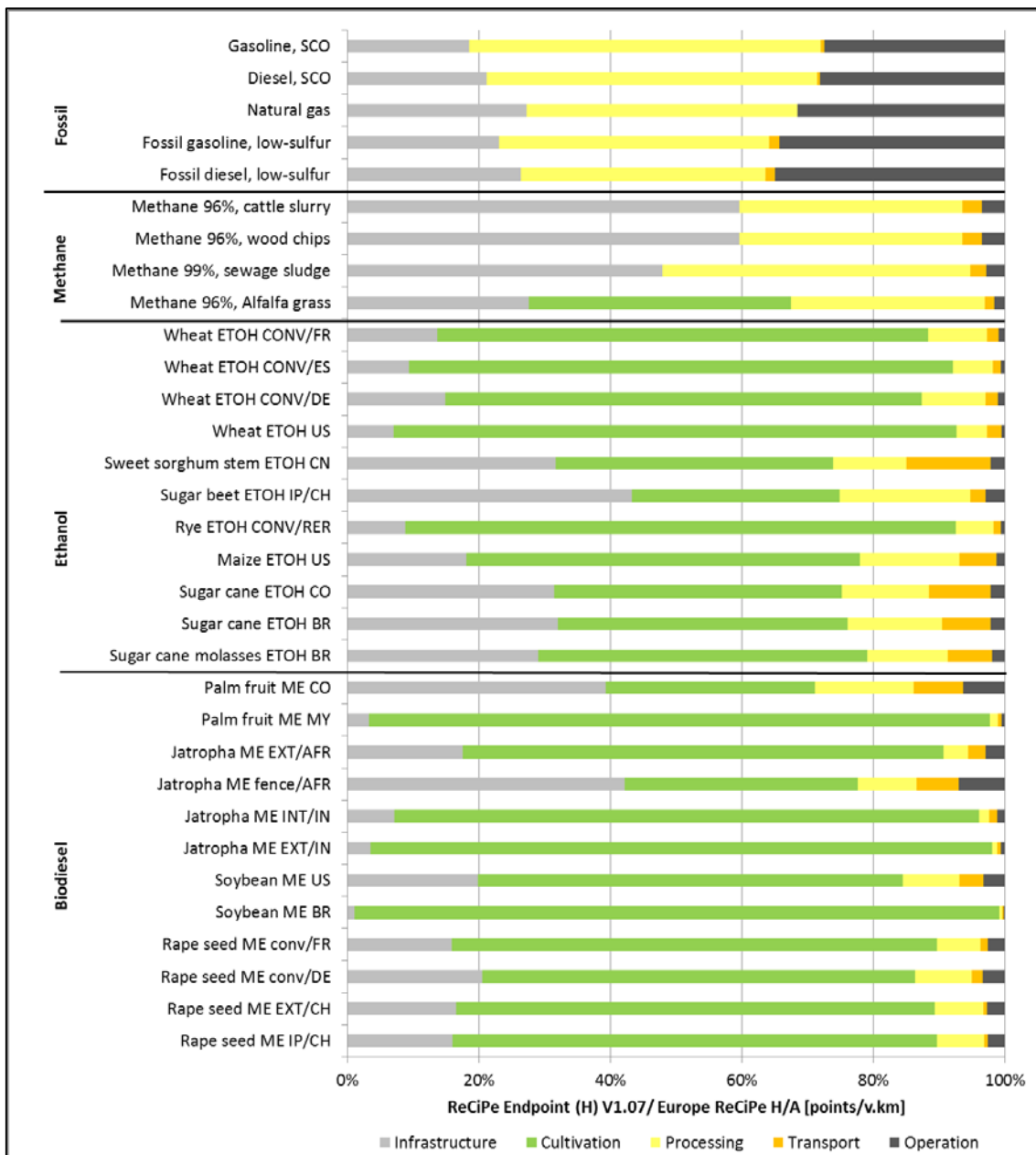

Abbildung 1: **Bedeutung der Lebensphasen der Biotreibstoffketten, berechnet mit der Wirkungsabschätzungsmethode UBP (2006).**

Abbildung 2: Bedeutung der Lebensphasen der Biotreibstoffketten, berechnet mit der Wirkungsabschätzungsmethode ReCiPe 2008.

Wie unterscheiden sich die aktuellen Resultate von der Studie 2007?

Für die meisten Biotreibstoffketten sind die Resultate stabil geblieben. Die Neuberechnung der Stickstoffemissionen führt im Allgemeinen zu einer Reduktion der Lachgasemissionen und damit zu einer geringeren Treibhausgas-Belastung. Die Aktualisierung der Nitrat- und Ammoniakemissionen zeigt dagegen keinen eindeutigen Trend, während die Neuberechnung der Landumwandlungsprozesse zu signifikant höheren Treibhausgas-Emissionen führt.
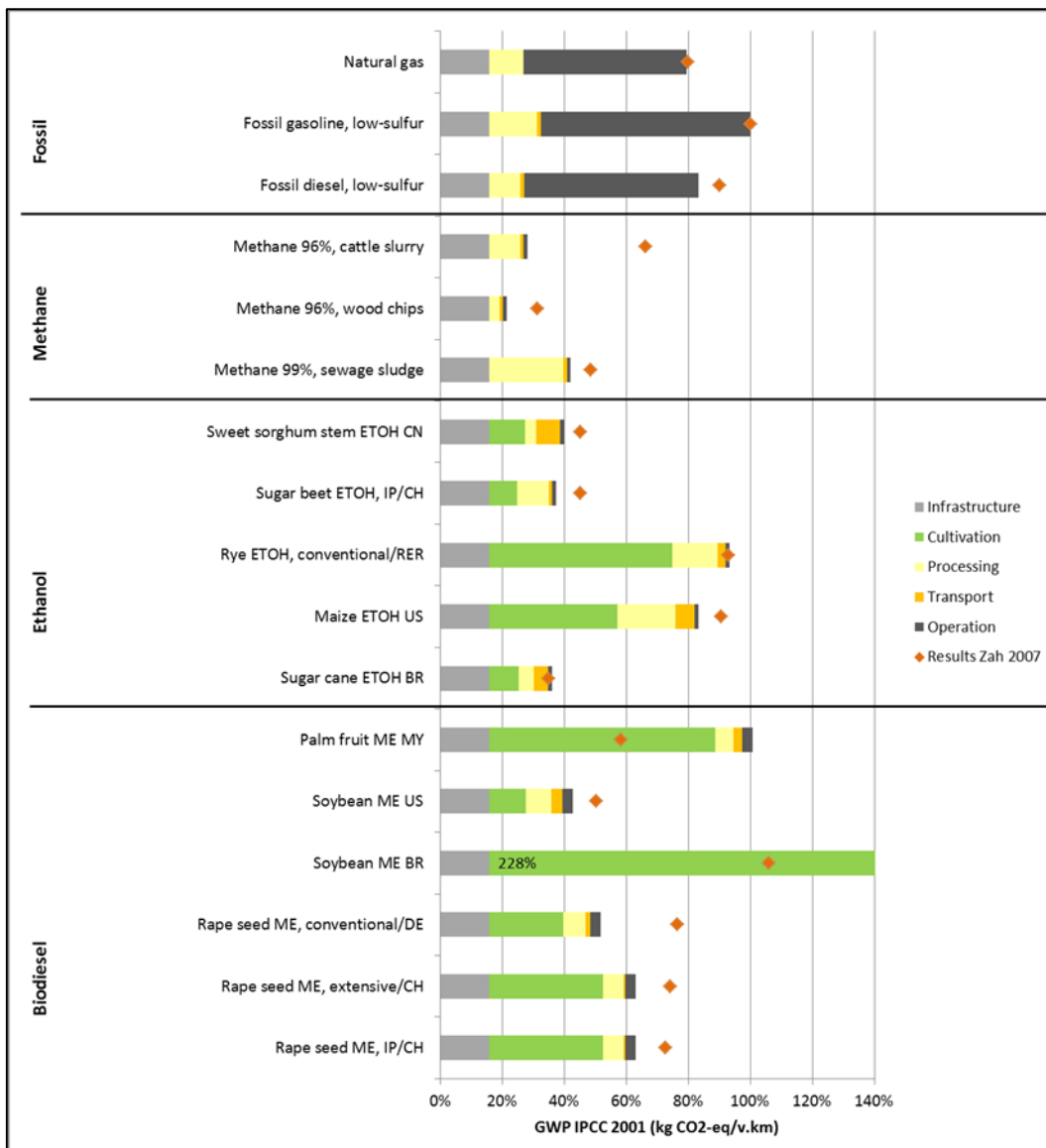
Abbildung 3: **Relativer Vergleich der GWP IPCC 2001 Resultate für Personentransport in Mittelklasswagen mit EURO3 Emissionsstandard (in kg CO$_2$-eq/v.km) der 2007 Studie und der aktualisierten Inventare der gleichen Produktionspfade als 2007. Fossiles Benzin stellt 100% der 2007 und der aktualisierten Resultate dar. Die Resultate für Sojabohnen ME aus Brasilien sind insgesamt 228% der Referenz.**

Die neuen SNG-Prozesse zeigen tiefere Umweltwirkungen als in der Studie 2007. Dies wird durch die im Vergleich zur Studie von 2007 tieferen Methanemissionen bei der landwirtschaftlichen Vergärung verursacht.

Welche Tendenzen zeigen sich für die fossile Referenz?

Weil man davon ausgeht, dass der Erdöl-Mix in Zukunft immer mehr unkonventionelle Quellen wie Ölsande oder Schiefergas enthalten wird, wurde in dieser Studie auch die Frage untersucht, wie sich die fossile Referenz in Zukunft entwickeln könnte. Aus Gründen der Datenverfügbarkeit und der zunehmenden Bedeutung dieser Rohstoffe auf dem globalen Markt wurden Treibstoffe aus kanadischem Ölsand analysiert, auch wenn diese gegenwärtig nicht im Schweizer Mix enthalten sind.

Die Resultate für Ölsande zeigen deutlich, dass die aufwändigere Produktion sich auf die Treibhausgasemissionen und auf die fossilen Energie- und Wasserverbräuche niederschlägt. Zudem ergeben sich auch signifikant höhere Umweltbelastungen für Eutrophierung, Umwandlung von Naturflächen, Partikelbildung und Human-Toxizität. Und dies obwohl

Auswaschungen aus den Sammelbecken der Förderrückstände, die ein hohes Umweltrisiko darstellen, mangels zuverlässiger Datengrundlage nicht berücksichtigt werden konnten. Man kann erwarten, dass sich die Umweltauswirkungen der Ölsand-Förderung dadurch noch deutlich erhöhen.

Grundsätzlich würde eine Beimischung unkonventioneller Öle die Umweltbelastung des Schweizer-Erdöl-Mixes erhöhen. Es muss hierbei die Frage beantwortet werden, ob die Referenz für die Bewertung der Biotreibstoffe diesem Trend folgen soll, mit der Konsequenz, dass die Biotreibstoffe immer tieferen Umweltstandards genügen müssten.

Wie beeinflusst die Wahl der Bewertungsmethode die Schlussresultate?

Die Verwendung von Midpoint-Methoden ermöglicht die präzise Ursachen-Analyse der verschiedenen Umweltbelastungen. So lässt sich das Treibhauspotenzial auf die emittierten Treibhausgase oder die Eutrophierung auf den erfolgten Nährstoff-Eintrag zurückführen. Midpoint-Indikatoren erlauben aber wegen der grossen Anzahl an Einzelindikatoren und den Trade-offs zwischen ihnen keine eindeutige Gesamtaussage. Deshalb ist es ebenfalls wichtig, die Analyse auf Endpoint-Ebene zu ergänzen.

Endpoint-Ansätze beruhen auf unterschiedlichen Konzepten und Modellannahmen. Deshalb können die Resultate verschiedener Bewertungsmethoden für die gleichen Produktionsketten zu sehr unterschiedlichen Resultaten führen. Die in der Schweiz entwickelte Methode der ökologischen Knappheit (UBP 2006) reflektiert Emissionsgrenzwerte und politische Umweltziele der Schweiz, während EcoIndicator 99 und die Nachfolger-Methode ReCiPe 2008 auf Effekt-Wirkungs-Modellen beruhen. Entsprechend gewichtet die UBP 2006-Methode z.B. Nitrat- und Schwermetallemissionen stark, während die ReCiPe 2008-Methode aufgrund der Modellierung der Kosten des Abbaus fossiler Ressourcen und der Wertung der Klimaproblematik diesen zwei Umweltauswirkungen ein grosses Gewicht gibt. Dies führt dazu, dass die meisten Biotreibstoffe bei der Bewertung mit der ReCiPe-Methode tendenziell günstiger abschneiden als mit der UBP-Methode.

Wie gross sind die Unsicherheiten der Resultate?

Die Unsicherheiten wurden auf Stufe der Inventarflüsse mit Monte Carlo Analysen untersucht. Die Resultate sind je nach Indikator unterschiedlich, im Allgemeinen liegt der Variationskoeffizient des Mittelwerts zwischen 10% und 50%. Für einen wichtigen Indikator wie das Treibhausgaspotential ist die Unsicherheit mit einem Variationskoeffizienten des Mittelwerts von ca. 30% relativ hoch, weil er von unsicheren Faktoren wie Landumwandlung oder Lachgasemissionen abhängt.

Fazit

Biotreibstoffe können eine Reduktion des fossilen Energieverbrauchs und der Treibhausgasemissionen ermöglichen. Sie führen aber oft auch zu einer Verlagerung der Umweltauswirkungen und generieren dadurch neue Umweltprobleme. Die aus Umweltsicht besten Produktionspfade basieren auf der Methanisierung von Bioabfällen und auf Biomasse aus nachhaltiger Waldbewirtschaftung.

Die Studie bestätigt die hohe Variabilität in den Umweltbelastungsprofilen der Biotreibstoff-Lebenszyklen und damit die Notwendigkeit, einzelne Projekte mit spezifischen Daten zu untersuchen.

Die Beimischung von Treibstoffen aus unkonventionellen fossilen Quellen würde die Umweltauswirkungen des Schweizer-Erdöl-Mixes tendenziell erhöhen. Sollte die fossile Referenz bei der Bewertung von Biotreibstoffen diesem Trend folgen, hätte das zur Folge, dass die Biotreibstoffe immer tieferen Umweltstandards genügen müssten

Die Unsicherheit der Resultate ist generell hoch. Besonders für die Emissionen von landwirtschaftlichem Lachgas und für die Bilanzierung der indirekten Landumwandlung sind spezifischere Modelle nötig, da beide Faktoren einen grossen Einfluss auf die Endresultate haben können. Diese Unsicherheiten sollten zu einer allgemeinen Vorsicht bezüglich der Förderung von Biotreibstoffen führen.

x

Abbreviations

AGB	Above Ground Biomass
bbl bit	barrel of bitumen
BCM	biomass-CO_2-methane
BGB	Below Ground Biomass
CHP	Combined Heat and Power
CO_2-eq	Carbon dioxide equivalent
DM	Dry Matter
DOM	Dead Organic Matter
EI 99	Eco-Indicator 99
FFB	Fresh Fruit Bunch
GHG	Greenhouse Gas
GWP	Global Warming Potential
ICE	Internal Combustion Engine
ILCD	International Reference Life Cycle Data System
iLUC	Indirect Land Use Change
JME	Jatropha Methyl Ester
JS	Jatropha seeds
LCA	Life Cycle Assessment
LCI	Life Cycle Inventory
LCIA	Life Cycle Impact Assessment
LUC	Land Use Change
PM	Particulate Matter
RED	Renewable Energy Directive
RER	Europe
SALCA	Swiss Agricultural Life Cycle Assessment
SB	Soybean
SC	Sugarcane
SCO	Synthetic Crude Oil
SRC	Short Rotation Coppice
SNG	Synthetic Natural Gas
SOC	Soil Organic Carbon
SQCB	Sustainable Quick Check for Biofuels
UBP	Umweltbelastungspunkte
v.km	Vehicle*kilometer

Table of content

1. Introduction

Life Cycle Assessment (LCA) has become in the last years in Switzerland but also on an international level an important assessment tool for bioenergy and has been included in several certification schemes. The availability of public and reliable data is a central precondition to support such assessments. However, the growing bioenergy sector is highly dynamic. On one hand, new energy crops, agricultural practices and improved processing technologies affect the environmental footprint of biofuels. On the other hand, the methods for measuring material flows and emissions as well as evaluating their environmental impacts get continuously improved. Consequently, it is a key issue to build on the current state of the art of biofuels production, when benchmarking the sustainability of biofuels. In this project we therefore aim at improving, harmonizing and completing the ecoinvent datasets pertaining to bioenergy as well as to evaluate the effects on this update on the assessment of biofuel pathways..

Inventories of biomass production and agricultural processes are one focus of this project. The modelling of nitrogen emissions as important contributors to the greenhouse gas emissions as well as of other environmental impacts like eutrophication and acidification was harmonized and updated. Furthermore, we developed a unified method to account for the emissions due to land use change, which have a significant influence on the climate change potential of the bioenergy production pathways (Reinhard, J. 2007; Zah, R. *et al.* 2007; Fargione, J. *et al.* 2008; Schmidt, J.H. 2008; Wicke, B. *et al.* 2008), whereas in ecoinvent v2.2 only the emissions due to cutting of rainforest had been considered,. An additional focus of this project was the extension of the ecoinvent database with inventories for Jatropha and grass mixtures.

A second focus of this project is the **extension of inventories on the conversion step** from feedstocks to biofuels as well as inventories for the production of heat and electricity. Specifically, hydrothermal gasification of waste material from agriculture and BCM-process for the purification of raw gas to methane were addressed in this project. Biomass waste ends partly as municipal waste and is incinerated in municipal waste incinerators, which also produce usable heat and electricity from the incinerated waste. For the better assessment of the optimal treatment and recovery of biomass waste options, the ecoinvent datasets of waste incineration plants were updated and refined. The recovering of oil sand from Canada as an example for unconventional oil sources was also studied in this project. This allows a first assessment of the trends in the environmental impacts of fossil fuel in future.

The third focus was an **update of the biofuel LCA study** (Zah, R. *et al.* 2007), which this extension and update of the ecoinvent bioenergy inventories allows. This study performs an environmental assessment of the new crop inventories and conversion processes; where needed we use the v2.2 inventories to complete the value chains.

This report is structured around these focuses of the project. In the **chapter 2**, we describe the update and harmonization of the agricultural inventories as well as of the accounting of land use change emissions. **Chapter 3** focuses on the inventories around the conversion step and on the biowaste combustion in municipal waste incinerators. **Chapter 4** evaluates the new and the updated biofuel pathways in comparison to different fossil references and discusses the results.

2. Agricultural processes

2.1. Methods

2.1.1. Emissions of ammonia to the air

Ammonium (NH_4^+) contained in fertilisers can easily be converted into ammonia (NH_3) and released to the air. Agriculture is the biggest source of ammonia emissions in Switzerland. For 2000, Thöni *et al.* (2007) estimated the total emissions of NH_3 to be 53,000 tonnes, 93% thereof from agriculture. Animal husbandry (emissions in the stable, during manure storage and spreading) is the largest source. About 30% of the excretions of N are lost in the form of ammonia. By taking appropriate measures, these emissions could be reduced by about 20-40% (Menzi, H. *et al.* 1997).

Ammonia contributes to acidification and the eutrophication of sensitive ecosystems. Its impact is mainly local and regional.

The emissions of NH_3 were calculated based on the model Agrammon (www.agrammon.ch), a model especially designed for the assessment of NH_3 emissions from agriculture on either farm scale or on a regional scale. The relevant modules of the model applied here are on the farm scale. The module 'application' refers to emissions from the application of farm manure and the module 'plant production' refers to emissions from the application of mineral and recycling fertilisers. The model structure and technical parameters can be found in Agrammon Group (Agrammon group 2009a, b). The implementation of the Agrammon emission factor in the ecoinvent inventories is described in (www.ecoinvent.org/talkpages).

The main changes compared to ecoinvent v2 are in the emissions from application of farmyard manure, whereas for the emissions after application of mineral N fertilizers, the same factors have been used.

2.1.2. Nitrate Leaching to Ground Water

Nitrate (NO_3^-) is either supplied to the soil by fertilizers or produced by micro-organisms in the soil via the mineralization of organic matter. Nitrate in the soil can be absorbed as a nutrient by the plants. In periods of heavy rainfall, however, precipitation exceeds soil evaporation and transpiration of the plants, which leads initially to saturation of the soil with water, and afterwards to percolation to the ground water. As nitrate is easily dissolved in water, the risk of leaching is high.

The risk of nitrate leaching is highest in autumn and winter, when precipitation often or always exceeds uptake by the plants. Moreover, nitrogen mineralisation is generally highest in late summer, when the nitrogen often cannot be taken up by the plants (Stauffer, W. *et al.* 2001).

Experiments have shown that it is not the choice of crops but rather the succession of crops in a crop rotation that is determining the amount of nitrate leached (Stauffer, W. *et al.* 2001). Since the modules in the ecoinvent database are life cycle inventories of products taking into account one single crop only, the succession of crops can only partly be taken into account. This fact should be borne in mind when interpreting the nitrate leaching values.

Nitrate losses are undesirable for several reasons:

–   From the agricultural point of view, valuable nutrients are lost from the soil, increasing the need for fertilisers.

–   Nitrate in ground water used as drinking water may have a toxic impact on humans. Although the acute toxicity of nitrate is low, nitrate is easily converted into nitrite, which has a higher acute toxicity and is supposed to be indirectly carcinogenic (Surbeck, A. *et al.* 1998).

–   Once ground water becomes surface water, nitrate contributes to eutrophication and also induces emissions of nitrous oxide, a major greenhouse gas (Schmid, M. *et al.* 2000).

Depending on the country of crop production different models were used to calculate nitrate leaching. A model by Richner *et al.* (in prep.) specifically for the application to conditions in Switzerland (SALCA-NO3) was applied to Switzerland and other European countries, where

similar conditions are found. For non-European countries the SQCB-NO3 model was used, a geographically unspecific and simplified model (De Willigen, P. 2000; Faist Emmenegger, M. *et al.* 2009).

The SALCA-NO3 model

Geographic scope of application: Europe

The model SALCA-NO3 calculates the expected nitrate leaching and comprises the following elements (Richner, W. *et al.* in prep.):

  – Nitrogen mineralisation from the soil organic matter per month
  – Nitrogen uptake by vegetation (if any) per month
  – Nitrogen input from the spreading of fertiliser
  – Soil depth
  – Factors not considered:
      o Amount of seepage
      o Denitrification

The use of the model in ecoinvent is described in (www.ecoinvent.org/talkpages).

The main changes of the model are the following:

  – The mineralization of N is made dependent on the region and the temperature.
  – The uptake of N by the vegetation was modeled by STICS (Brisson, N. *et al.* 2003).
  – Several model parameters were revised, based on new experimental data.

The SQCB-NO3 model

The SQCB-NO3 model is reported in Faist Emmenegger *et al.* (2009) and is an adaption of a formula developed by de Willigen (2000). The formula calculates the leaching of NO3-N and is a simple regression model of the form:

$$N = 21.37 + \frac{P}{c*L}\left[0.0037*S + 0.0000601*N_{org} - 0.00362*U\right]$$

where:

N = leached NO3-N [kg N/(ha*year)]

P = precipitation + irrigation [mm/year]

c = clay content [%]

L = rooting depth [m]

S = nitrogen supply through fertilisers [kg N/ha]

N_{org} = nitrogen in organic matter [kg N/ha]

U = nitrogen uptake by crop [kg N/ha]

It must be mentioned that in Faist Emmenegger *et al.* (2009) the formula has been taken from Roy *et al.* (2003), where it is not reported correctly (p. 51, formula "OUT3"), stating C_{org} instead of N_{org}.

The SQCB model provides relatively simple approaches to assess most of the required input parameters. P and C_{org} are determined through the ecozone in which the crop is produced.

In case of irrigation, the amount of irrigation water [mm] is added to the precipitation in order to obtain the parameter P. The detailed implementation of the model in ecoinvent is described in (www.ecoinvent.org/talkpages).

2.1.3. Emissions of N_2O to the air

Nitrous oxide or dinitrogen monoxide (N_2O) is produced as an intermediate product in the denitrification process (conversion of NO_3^- into N_2) by soil micro-organisms. It can also be produced as a by-product in the nitrification process (conversion of NH_4^+ into NO_3^-, (Schmid, M. *et al.* 2000)). The total emissions of N_2O caused by the Swiss agricultural sector in 1996 were estimated at 8,600 tonnes. N losses in the form of N_2O are closely linked to the nitrogen

cycle in agriculture; intensive agriculture with a high input of nitrogen fertiliser contributes to the increase in N_2O-emissions. N_2O is a greenhouse gas with a high impact.

Calculations of N_2O emissions are based on the IPCC method for calculating N_2O emissions (Eggleston, H.S. *et al.* 2006). Direct emissions of N_2O and indirect or induced emissions are included. In the case of indirect N_2O emission, nitrogen is first emitted as NH_3 or NO_3^- and subsequently converted to N_2O.

Direct N_2O emissions [kg N_2O] from mineral and organic fertilisers and from crop residues were calculated on the basis of the total nitrogen content (N_{tot} [kg N]). The factor of 1.0% N lost as N_2O was used. Induced N_2O emissions [kg N_2O] from ammonia (NH_3) were considered using a factor of 1.0% of N in emitted NH_3, induced emissions from nitrate (NO_3^-) with a factor of 0.75% of N in leached NO_3^-. According to the new IPCC-guidelines (IPCC 2006), no emissions are calculated from biological nitrogen fixation.

The content of total nitrogen in farmyard manure was taken from Walther *et al.* (2001).

The contents of total nitrogen in crop residues are taken from Walther *et al.* (2001) and the amounts of crop residues from Nemecek *et al.* (2007) as such, or scaled by the yields of the reference products (and by-products) based on the latter.

Updated model:

N_2O = $44/28 * (0.01 (N_{tot} + N_{cr}) + 0.01 * 14/17 * NH_3 + 0.0075 * 14/62 * NO_3^-)$

N_2O = emission of N_2O (kg N_2O/ha)

N_{tot} = total nitrogen in mineral and organic fertilisers (kg N/ha)

N_{cr} = nitrogen contained in the crop residues (kg N/ha)

NH_3 = losses of nitrogen in the form of ammonia (kg NH_3/ha)

NO_3^- = losses of nitrogen in the form of nitrate (kg NO_3^-/ha).

Compared to the previous version, based on IPCC (Houghton, J.T. *et al.* 1996) & (IPCC 2001) the model has undergone substantial changes with a strong influence on the emissions. These changes are therefore highlighted.

Model used in ecoinvent v2:

N_2O = $44/28 * (\mathbf{0.0125 (N_{av}-14/17*NH_3+ N_{cr}+0.6N_{bf})} + 0.01 * 14/17 * NH_3 + \mathbf{0.025}*14/62 * NO_3^-)$

N_{av} = available nitrogen (kg N/ha)

The changes are printed in bold:

  – The general emission factor has been reduced from 1.25% to 1.0%. This factor applies to the inputs of nitrogen as fertilisers and as crop residues. This change implies a 20% reduction of N2O emissions. Several reviews of new experimental results lead to an average emission factor close to 0.9%. Therefore 1.0% has been adopted as the new emission factors. We have to bear in mind that N2O emissions are highly variable and depend on a number of factors, which are not considered in the model. (IPCC 2006) gives an uncertainty range of 0.3% to 3%.

  – In ecoinvent v2, the available nitrogen has been used to calculate the N amounts in organic fertilisers. According to (IPCC 2006), the total N is used now. This change has no effect on the emissions from mineral fertilisers, since the total amount of nitrogen is assumed to be available. However, in situations where organic fertilizers are applied, this can increase the emissions.

  – According to (IPCC 2006) the emissions of ammonia are no more subtracted from the total amount of supplied nitrogen. (IPCC 2006) give the following explanation: „The reason for this change is that field studies that have determined N_2O emission factors for applied N were not adjusted for volatilization when they were estimated." Depending on the amount of ammonia loss, this can lead to an increase of N_2O emissions.

  – The effect of symbiotic nitrogen fixation (SNF) is no more considered as contributing to N_2O emissions according to (IPCC 2006). Experiments have shown that N_2O

4

emissions from crops with SNF are similar to crops without N fertilization. This leads to much lower emissions for legumes.
– Finally, the induced emissions from nitrate leaching have been reduced from 2.5% to 0.75%. This is a very significant reduction of more than a factor 3! In all situations, where nitrate leaching played an important role, this should lead to a substantial reduction of N_2O emissions. As the amounts of nitrate leaching can be considerable, for some crops this process made a significant contribution to the overall N2O emissions. We have to emphasize that this emission factor is particularly uncertain.

2.1.4. LUC emissions

Land use activities in the agro forestry sector are one of the main sources for anthropogenic greenhouse gas (GHG) emissions. The most important GHG emissions of concern are CO_2, N_2O (di-nitrogen monoxide) und CH_4 (methane) (IPCC 2006). Approximately 30% of all anthropogenic GHG emissions between 1989 and 1998 could be allocated to land use activities (Carmenza, R. *et al.* 2008). Land use changes (LUC), i.e. the transformation of one land use type to another, is responsible for approximately 2/3 of those emissions (Carmenza, R. *et al.* 2008). In this context, optimization of land use activities and in particular land transformations plays a key role in reducing GHG emissions.

The goal of this sub-module was the update of the emission from direct LUC for all relevant crop activities. The attribution of LUC associated with the increase in area of the relevant crop activities was derived from recent publications and data from the Food and Agriculture Organization of the United Nations (FAOSTAT) from 1990-2009. That is, for the consideration of LUC we applied a time period of 20 years. This is in line with PAS 2050 (BSI 2011) and recommended as a reasonable time period for the consideration of emissions from LUC (Brandão, M. *et al.* 2012; Koellner, T. *et al.* 2012; Milà i Canals, L. *et al.* 2012). In order to smooth out short-term fluctuations we used five year averages.

In order to determine the crop activities for which direct LUC is potentially of importance we applied the procedure of (Milà i Canals, L. *et al.* 2012) for a first screening. For each crop activity in ecoinvent, we analysed if (i) the crop area in the country and (ii) its corresponding total land type area has increased in the last 20 years. Since we focus only on the LUC from natural ecosystems, we analysed in addition if (iii) the natural ecosystem decreased during the same time period, i.e. if forest land or other land was affected by the increase in (i) and (ii). Direct LUC was considered to be potentially relevant only if all three conditions were met by an analysed crop activity. We used data from Food and Agriculture Organization of the United Nations (FAOSTAT) from 1990-2009 for this assessment.

According to this procedure direct LUC is potentially relevant (and hence further analysed) with regard to the following crop activities:
– Soybeans, at farm, BR (SB);
– Sugarcane, at farm, BR (SC);
– Palm fruit bunches, at farm, MY (FFB).

Almost all other existing ecoinvent crop activities related to bioenergy production are located in developed countries where arable land areas are constant and forest areas are slightly increasing. The remaining crop activities located in China show the same pattern.

For the newly established crop cultivation datasets, crop specific information based on field visits, combined with literature values and expert interviews were used to determine the former land use pattern of the respective crop. By using the best available and crop specific information, the consideration of LUC emissions turned out to be relevant for following new inventories:
– Palm fruit bunches, at farm, CO (FFB)
– Jatropha seeds, at farm, IN (JS)
– Jatropha seeds, at farm, AFR (JS)

The update focused on the consistent consideration of all carbon pools given by the (IPCC 2006). Three kinds of carbon pools are distinguished: above ground biomass (AGB) and below ground biomass (BGB), dead organic matter (DOM) and soil organic carbon (SOC). Figure 1 shows how these carbon pools were considered in the inventory modelling.
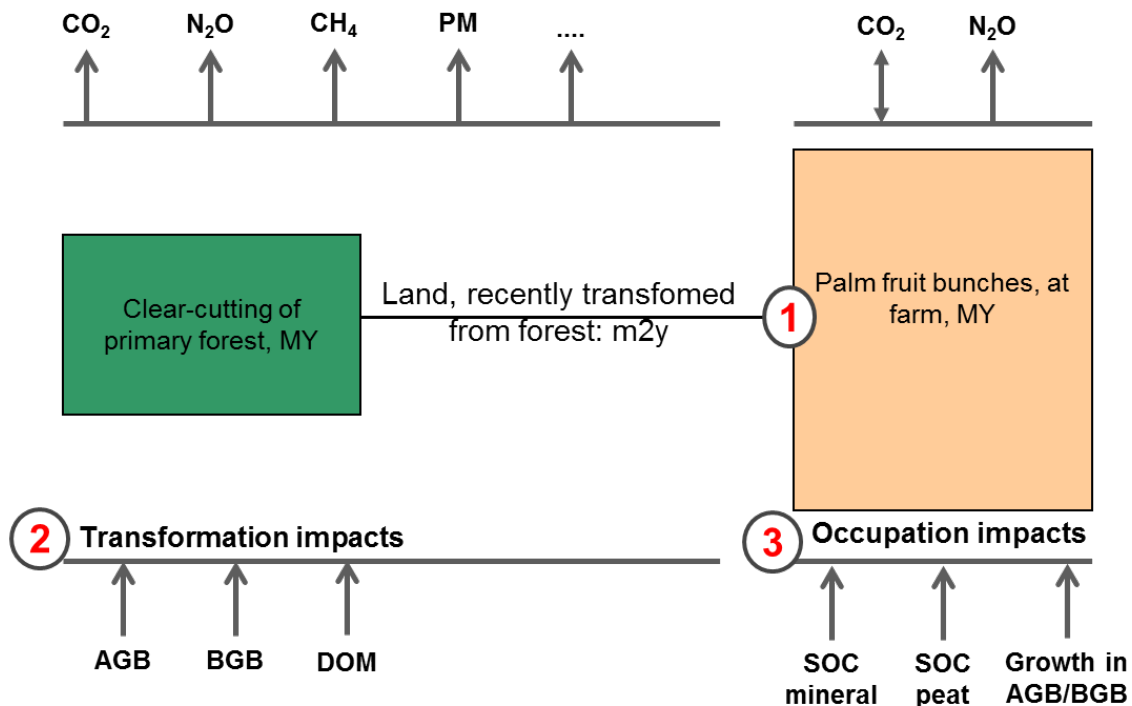
Figure 1:   Consideration of carbon pools in the inventory modeling (exemplary shown for Palm fruit bunches, at farm, MY).

As shown in Figure 1, the interventions were separated according to transformation and occupation impacts. In order to determine both impacts the LUC associated with the cultivation for each of the relevant crops is required (1). Transformation impacts (2) refer the interventions directly associated with the provision of the land from natural ecosystems, i.e. the emissions resulting from the loss of AGB, BGB and DOM. Occupation impacts (3) refer to the interventions directly associated with the use of the land, i.e. the loss/gain of SOC in mineral soil, the loss of SOC stored in organic peat soils and the permanent accumulation of carbon in AGB and BGB on the occupied land.

Assessment of land use change

Soybeans (SB)

The increase of the area, which is used for soybean cultivation in Brazil, is one important driver for the deforestation of the Amazon (Macedo, M.N. *et al.* 2012). Since 1990 the area cultivated with soybean has more than doubled from approximately 11 to 23 million hectares. According to (Macedo, M.N. *et al.* 2012), the expansion of the soybean cultivation area from 2001 to 2005 took place on rain forest land (26%) and shrubland (74%) and from 2005-2009 mainly on shrubland (91%). Using these shares and data from (FAOSTAT 2012) we calculated the area of soybean standing on rain forest and shrublands today (Figure 2).
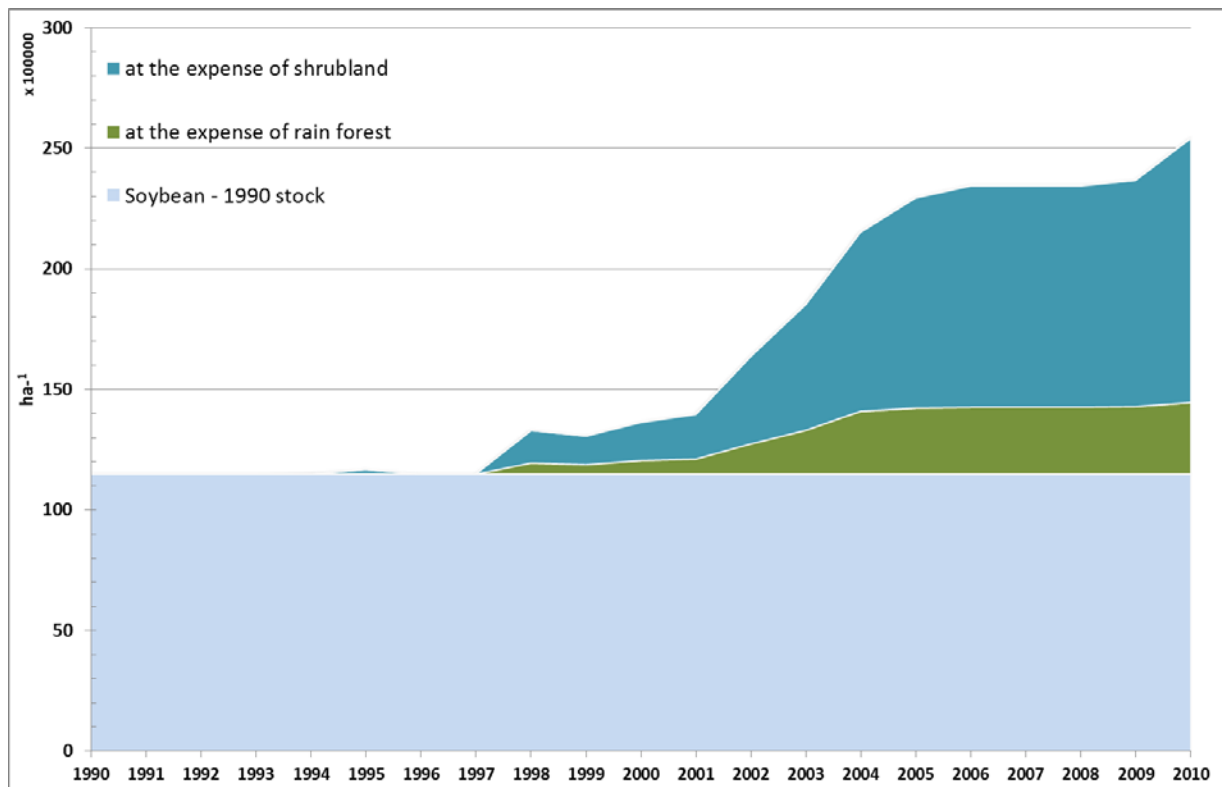
Figure 2: **Increase of soybean cultivation area [in 100'000 ha] in Brazil from 1990-2010 (FAOSTAT 2012).The increase at the expense of rain forest and shrubland is calculated on the basis of (Macedo, M.N. *et al.* 2012).**

Using 1990 as a baseline, i.e. a time horizon of 20 years, and smoothing out short term fluctuation by using 5 year averages, the average ha soybeans occupies 59% already cultivated land, 15% rain forest and 43% shrublands (see Figure 2).

The associated land transformations are annualized using the applied time period (20 years) and modelled by means of the activities "land, recently transformed from primary forest, BR" (PF) and "land, recently transformed from shrublands, BR" (SH) which are explained in detail in the section Transformation impacts.

Palm fruit bunches, Malaysia (FFB)

The increase of the oil palm cultivation area in Malaysia is an important driver for the deforestation of the Malaysian rain forest (Reinhardt, G. *et al.* 2007). Since 1990 the area cultivated with oil palms has more than doubled from approx. 1.7 to 4 million hectares. According to (Reinhardt, G. *et al.* 2007), the increased palm oil production caused an expansion into rain forest lands and replaced natural rubber, coconut and cocoa plantations. The difference between the decrease in the mentioned plantations and the increase in area cultivated with oil palm is assumed to occur fully at the expense of rain forest. Using data from (FAOSTAT 2012) from 1990-2009 we calculated the area of oil palm standing on rain forest today (Figure 3).
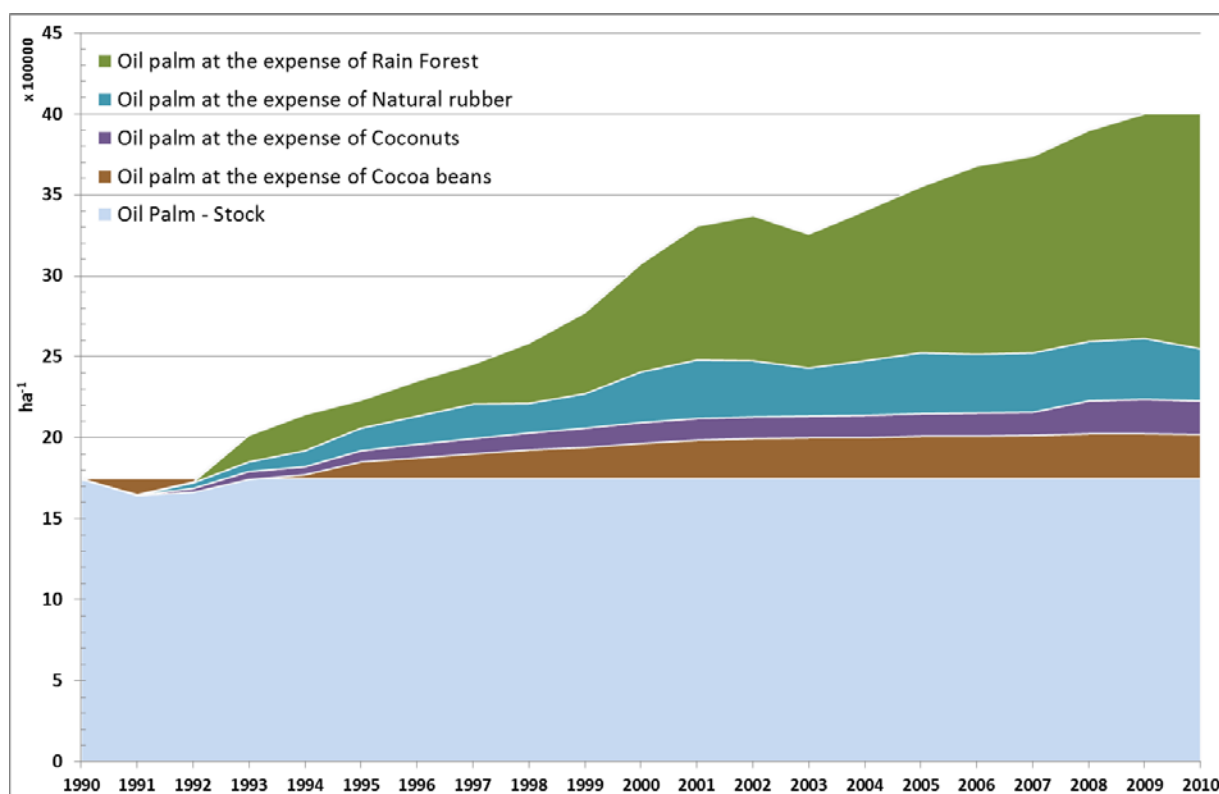
Figure 3: **Increase in area cultivated with oil palm from 1990-2010 (FAOSTAT 2012). The increase at the expense of rain forest is calculated on the basis of (Reinhardt, G.** *et al.* **2007).**

Using 1990 as a baseline, i.e. a time horizon of 20 years, and smoothing out short term fluctuation by using 5 year averages, the average ha FFB occupies 51% already cultivated land, 21% other plantations and 28% rain forest (or 1'071'138 ha); an amount that is rather at the lower end of the estimated area deforested due to FFB in this time period (Grieg-Gran, M. *et al.* 2007). The share of primary forest corresponds to ~1 million ha. Roughly half of FFB on rain forest is on peat land (Page, S.E. *et al.* 2011).

The associated land transformations are annualized using the applied time period (20 years) and related to the activity by means of the activity "land, recently transformed from primary forest, MY". No land transformation is considered for the mentioned plantations given that they refer to the same land use category, i.e. permanent crops.

Palm fruit bunches, Colombia (FFB)

According to FAO, the Colombian oil palm cultivation area expanded from 1990 to 2009 by 84% (FAOSTAT). According to Fedepalma, the expansion of oil palm plantation since 1990 caused a conversion of shrubland (50.7%), grassland (41.5%), arable land (6.8%), peatland (0.6%) and primary forest (0.4%).

Sugarcane (SC)

For Sugarcane no justification was found for an update of the assumption from 2007, i.e. that 1% of sugarcane cultivation in Brazil goes at the expense of shrublands. Consequently, we added the inventory "land, recently transformed from shrublands"

Jatropha seeds, India (JS)

Since most *Jatropha curcas L.* plantations were established within the past few years, there is not yet an FAO statistic about jatropha available. However, the GEXSI study surveyed 242 jatropha projects in 55 countries (GEXSI 2008). According to GEXSI, the Asian jatropha plantations were established on wasteland (54%), agricultural land (42%), secondary forest land (4%) and primary forest land (0.4%). The "wasteland" category is very heterogeneous,

containing land ranging from woodland to highly eroded land. Within this study, the carbon stock of "wasteland" was approximated with the carbon stock of "grassland".

Jatropha seeds, Eastern Africa (EA)
According to (Mogaka, V. *et al.* in preparation), jatropha block plantations were only recently established, mainly on arable land (92.5%), grassland (4.1%) and shrubland (3.4%).

Table 1 summarizes the land transformation related to SB and FFB based on the prior analysis in comparison to ecoinvent v2.2.

Table 1: **Annual land transformation associated with the land use of one ha^{-1}*y^{-1} for SB and FFB, respectively. The share of peat land is part of the land on primary forest and thus not added to the total.**

Criteria	Unit	SB, v2.2	SB, v3	FBB, v2.2	FBB, v3	SC, v2.2 & v3
Yield	kg/ha^{-1}	2'544	2'544	24'978	24'978	68'305
Applied time period	yr^{-1}	2	20	25	20	20
Land on primary forest	m^2*yr^{-1}/ha^{-1}	158	75	400	142	-
-Land on peat land	*m^2*yr^{-1}/ha^{-1}*	-	-	-	*70*	-
Land on shrublands	m^2*yr^{-1}/ha^{-1}	262	217	-	-	5
Land already transformed	m^2*yr^{-1}/ha^{-1}	4'580	208	-	358	495
Total	m^2*yr^{-1}/ha^{-1}	**5'000**	**500**	**400**	**500**	**500**

Table 2 summarizes the land transformation related to jatropha and palm fruit CO.

Table 2: **Annual land transformation associated with the land use of one ha^{-1}*y^{-1} for new inventories.**

	Unit	Oil palm	Jatropha, intensive	Jatropha, extensive	Jatropha, extensive	Jatropha, fence
Origin		**CO**	**IN**	**IN**	**AFR**	**AFR**
Yield	kg/ha^{-1}	19'900	2'028	634	800	5'000[1]
Applied time period	yr^{-1}	20	20	20	20	20
Land on primary forest	m^2*yr^{-1}/ha^{-1}	1	2	2	-	-
Land on secondary forest	m^2*yr^{-1}/ha^{-1}	-	20	20	-	-
Land on peat	*m^2*yr^{-1}/ha^{-1}*	*1*	-	-	-	-
Land on shrubland	m^2*yr^{-1}/ha^{-1}	137	-	-	17	-
Land on grassland	m^2*yr^{-1}/ha^{-1}	113	267	270	21	-
Land on agriculture	m^2*yr^{-1}/ha^{-1}	18	211	208	463	-
Land on deadwood fences	m^2*yr^{-1}/ha^{-1}	-	-	-	-	500
Land already transformed	m^2*yr^{-1}/ha^{-1}	229	-	-	-	-
Total	m^2*yr^{-1}/ha^{-1}	**500.00**	**500.00**	**500.00**	**500.00**	**500.00**

---

[1] The average yield of a jatropha fence in East Africa is 1 kg per meter. The yield per hectare was calculated based on this value and the width of the fence (2meters).

Comparing SB v2.2 and v3, the updated land transformations are half for the transformation of primary forest and slightly smaller for the transformation of shrublands. The reason for these differences is that v2.2 extrapolated from the increase in 2003/2004: the time period with the highest increase in SB cultivation in Brazil (see Figure 4). For FFB the updated land transformation is lower by approx. 1/3 given that in v2.2 100% of the FFB was assumed to be cultivation on rain forest. The share of peat land (included in the land transformed from primary forest) was not considered in v2.2.

Figure 4 shows the share of the land use transformation from 1990 to 2009 for the different crops.
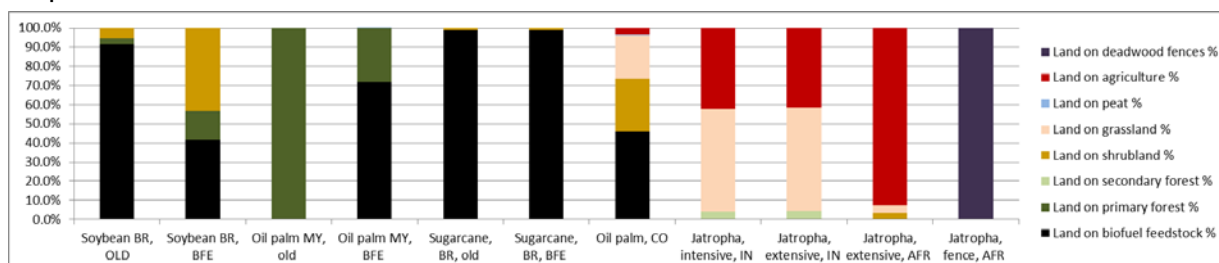
Figure 4: The relative land use transformation of the different biofuel feedstocks [%].

In the following we will highlight the main assumptions related to both the transformation and occupation impacts associated with the elaborated LUC.

Transformation impacts

As highlighted above, impacts associated with land transformation refer to the loss of AGB, BGB and DOM in the natural ecosystem. Arable land and deadwood fences are assumed to have no AGB, BGB and DOM stock (IPCC 2006) and thus no land transformation impact results. The biomass carbon stock of the land which was already under biofuel feedstock cultivation is assumed to remain the same. Figure 5 shows the general assumptions applied for the modelling of emissions related to land transformations from primary forests, secondary forests and shrubland.
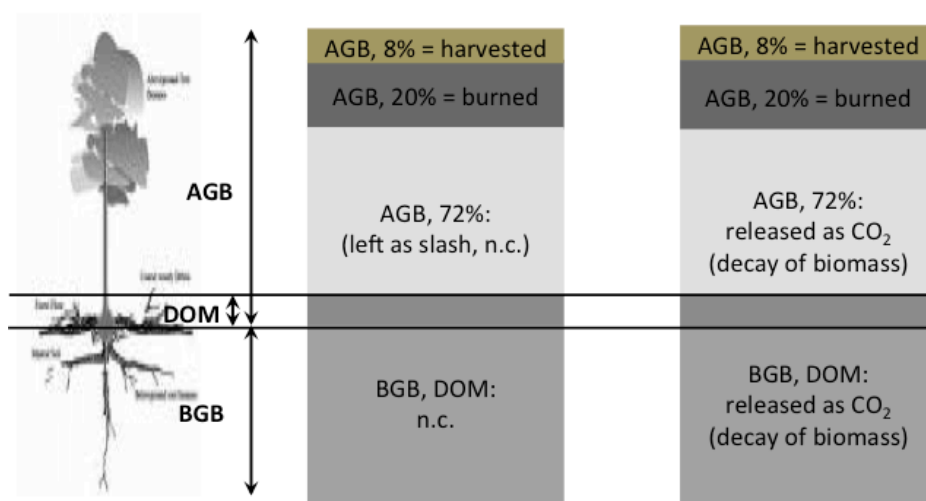
Figure 5: Assumptions for the modelling of emissions from the clear-cutting of primary and secondary forests and shrub lands for ecoinvent v2.2 (left) and v3 (right). Shares of AGB harvested and burned are from (Houghton, R.A. *et al.* 2000).

It is estimated that 20% of the AGB is burned and 8% harvested (Houghton, R.A. *et al.* 2000). In contrast to ecoinvent v2.2 it is assumed that the C stored in the remaining slash, the BGB and the DOM is released as CO_2. That is, the full amount of C stored in AGB, BGB and DOM not burned and harvested is transferred to the atmosphere in terms of CO_2. This is

in line with the default assumptions of the (2006). I2006). In the following, the biomass carbon stock of natural land of the specific countries is described.

For grassland no harvest or burning of biomass is considered. That is the full amount of AGB and BGB is transferred to the atmosphere in terms of CO_2. The values for AGB and BGB were taken from the IPCC (2006).

In the following, the biomass carbon stock of natural land of the specific countries is described and the new activity data is compared with the data used in ecoinvent v2.2.

Land transformation in Brazil

For Brazil, we updated the activity "land, recently transformed from primary forest" and created the new activity "land, recently transformed from shrublands". Data for the amount of AGB, BGB and DOM in primary forest were taken from (IPCC 2006). For the AGB and BGB content of shrublands we used the specific data provided by the (EU-Commission 2010). Table 3 shows the used carbon stock data in comparison with ecoinvent v2.2.

Table 3: **Carbon stocks for primary forest and shrub lands in Brazil and associated emissions of GWP (100a). Cells marked as n.c. indicate that this stock was not considered for the respective activity.**

Criteria	Unit	PF, v2.2	PF, V.3	SH, V.2.2	SH, V.3
AGB	kg C/ha^{-1}	190'000	141'000	n.c.	37'600
BGB	kg C/ha^{-1}	n.c.	52'170	n.c.	15'400
DOM	kg C/ha^{-1}	n.c.	7'600	n.c.	0
Total emissions	kg CO_2equiv./ha^{-1}	137'460	688'105	n.c.	196'624

As shown by Table 1, due to the consideration of the full amount of C stored in AGB, BGB and DOM not burned and harvested the emissions associated with the clear-cutting of one ha rain forest increase approx. by a factor 5 for the updated data. The clear-cutting of shrublands was not considered in v2.2.

Land transformation in Malaysia

For Malaysia, we updated the activity "land, recently transformed from primary forest". Data for the amount of AGB, BGB and DOM in primary forest were taken from IPCC (2006). Table 4 shows the used carbon stock data in comparison with ecoinvent v2.2.

Table 4: **Carbon stocks for primary forest in Malaysia and associated emissions of GWP (100a). Cells marked as n.c. indicate that this stock was not considered.**

Criteria	Unit	PF, V.2.2	PF, V.3
AGB	kg C/ha^{-1}	151'000	164'500
BGB	kg C/ha^{-1}	n.c.	60'865
DOM	kg C/ha^{-1}	n.c.	7'600
Total emissions	kg CO_2equiv./ha^{-1}	109'275	800'533

Land transformation in Eastern Africa

For Eastern Africa we created the activity "land, recently transformed from grassland" and "land, recently transformed from shrubland".

Table 5: Carbon stocks for grassland and shrubland in Eastern Africa and associated emissions of GWP (100a). n.r.: not relevant.

Criteria	Unit	Grassland, AFR	Shrubland, AFR
AGB	kg C/ha^{-1}	5'523	37'600
BGB	kg C/ha^{-1}	2'043	15'040
DOM	kg C/ha^{-1}	n.r.	n.r.
Total emissions	kg CO_2equiv./ha^{-1}	27'746	184'060

Land transformation in India

For India we created the activity "land, recently transformed from grassland", "land, recently transformed from secondary forest" and "land, recently transformed from primary forest".

Table 6: Carbon stocks for grassland, secondary and primary forest in India and associated emissions of GWP (100a). n.r.: not relevant.

Criteria	Unit	Grassland, IN	Secondary Forest, IN	Primary Forest, IN
AGB	kg C/ha^{-1}	5'523	106'000	131'600
BGB	kg C/ha^{-1}	2'043	39'100	48'692
DOM	kg C/ha^{-1}	n.r.	n.r.	3'650
Total emissions	kg CO_2equiv./ha^{-1}	27'746	506'793	629'736

Land transformation in Colombia

For Colombia we created the activity "land, recently transformed from grassland", "land, recently transformed from shrubland", "land, recently transformed from primary forest"

Table 7: Carbon stocks for grassland, secondary and primary forest in India and associated emissions of GWP (100a). n.r.: not relevant.

Criteria	Unit	Grassland, CO	Shrubland, CO	Primary Forest, CO
AGB	kg C/ha^{-1}	5'523	37'600	141'000
BGB	kg C/ha^{-1}	2'043	15'040	52'170
DOM	kg C/ha^{-1}	n.r.	n.r.	3'650
Total emissions	kg CO_2equiv./ha^{-1}	27'746	184'060	688'100

Occupation impacts

As highlighted in Figure 1, impacts associated with occupation regard (i) the loss/gain of SOC in mineral soil (and the related release of N_2O), (ii) the loss of SOC stored in organic peat soils and (iii) the accumulation of carbon in AGB and BGB on the new land use that is not harvested.

Regarding (i) the loss/gain in mineral soils is calculated using the factors for land use and management given by the IPCC (2006). The required reference carbon stock prevailing in the respective native vegetation is taken from (Brandão, M. *et al.* 2012). It is important to note that the losses in mineral soils are only considered for land transformed from native vegetation while gains in mineral soils are only considered for land transformed from land already in use.

With regard to (ii), tropical peat land is one of the Earths most spatially efficient carbon sinks. The cultivation of oil palm plantation on peat soils typically reduces or removes the carbon sink capacity of the peat land causing large carbon losses arising particularly form enhanced

aerobic peat decomposition (Page, S.E. *et al.* 2011). The annual emission factor from the transformation of peat soils to oil palm plantations is taken from (Page, S.E. *et al.* 2011).

For (iii) the increase in AGB and BGB on the new land use is not considered for land transformations within one land use category, e.g. for the transformation from permanent crops (cacao plantations) to permanent crops (oil palm plantations) no increase in AGB and BGB is considered. The increase is only relevant for permanent crops such like oil palm and jatropha. The data is taken from (EU-Commission 2010).

Table 8 shows the occupation impacts of SB, FFB and SC. Each value listed in Table 8 already refers to the respective land transformation values given in Table 1. For example, in a 20 year perspective, roughly 29 tons of carbon are emitted every year per ha peat land cultivated with oil palms (Page, S.E. *et al.* 2011). The average annual share of land transformed from peat land is ~14%. By multiplying the annual carbon loss/gain per hectare with this share we obtain the value of ~4'000 kg C given in Table 8.

Table 8: **Annual occupation impacts associated with the land use for SB & FFB ecoinvent v2.2 and v3. Negative values indicate accumulation. Cells marked as n.c. indicate that this stock was not considered, n.r. means not relevant.**

Criteria	Unit	SB, v2.2	SB, v3	FFB, v2.2	FFB, v3	SC, v3
SOC – mineral	kg C/ha^{-1} yr^{-1}	195	891	644	n.r.	n.r.
SOC – peat	kg C/ha^{-1} yr^{-1}	n.r.	n.r.	n.c.	4'059	n.r.
Gain in AGB/BGB	kg C/ha^{-1} yr^{-1}	n.r.	n.r.	n.c.	-851	-200
Total	kg C/ha^{-1} yr^{-1}	195	891	644	3'208	-200

As shown in Table 8, the most significant occupation impact is caused by oil palm cultivation in Malaysia, due to the drainage of peat land. In ecoinvent v2, neither the carbon emissions of draining peat land, nor the biomass carbon stock of the biofuel crop was considered. This has been updated consistently and also the new inventories of oil palm cultivation in Colombia and jatropha cultivation in Africa and India are based on the same methodology (Table 9). The new datasets generally show an occupation benefit, either due to the accumulation of biomass or/and due to the increase in SOC. The accumulation of SOC is an effect of the reduced tillage due to cultivating perennial crops on former agricultural land, which is especially the case for jatropha cultivation.

Table 9: **Annual occupation impacts associated with the land use of the new inventories. Negative values indicate accumulation. Cells marked as n.c. indicate that this stock was not considered.**

Criteria	Unit	Oil palm, CO	Jatropha, intensive, IN	Jatropha, extensive, IN	Jatropha, extensive, AFR
SOC – mineral	kg C/ha^{-1} yr^{-1}	-18.9	-270.95	-148.68	-330.60
SOC – peat	kg C/ha^{-1} yr^{-1}	69.11	n.r.	n.r.	n.r.
Gain in AGB/BGB	kg C/ha^{-1} yr^{-1}	-1293.23	-690.9	-690.9	-690.9
Total	kg C/ha^{-1} yr^{-1}	-1243.06	-961.85	-839.58	-1021.50

2.2. **New crops**

2.2.1. **Alfalfa-grass mixture, Switzerland**

Alfalfa-grass mixtures and so-called "Mattenklee"-grass mixtures are used for sown temporary leys. They can be used for cutting and not for pasture. One important characteristic is that these mixtures are able to fix high amounts of nitrogen. The inventory comprises a cultivation duration of 2.5 years. The meadow is sown in August and used during two subsequent years. The total yield over the cultivation period is 30'002 kg DM/ha.

The meadow is cut twice in the sowing year and four times per year in the following ones. 20 t of solid are applied before sowing, 20 m^3 liquid manure are applied per main year. The data come from (Nemecek, T. *et al.* 2005).

2.2.2. *Miscanthus* and *Salix*, Switzerland

The data for *Miscanthus* and *Salix* come from the study of (Hölscher, T. *et al.* 2007) and are valid for Southern Germany.

The cultivation period for *Miscanthus* is 20 years. The harvests starts in the second year (yield 6 t DM ha^{-1}) and are continued from the 3rd to the 20th year with yields of 17 t DM ha^{-1} year^{-1}.

The cultivation period of *Salix* (willow, short rotation coppice) is also 20 years. Harvests start in the 4th year (32 t DM ha^{-1}) and are continued in intervals of 3 years (7th year: 34 t DM ha^{-1}, 10th year: 38 t DM ha^{-1}, 13th year: 40 t DM ha^{-1}, 16th year: 34 t DM ha^{-1}, 19th year: 40 t DM ha^{-1}).

For the detailed description of the inventories, please consult the ecoinvent documentation.

2.2.3. Sugar cane cultivation, Colombia

The inventory for Colombian sugar cane cultivation is modelled based on field data from the main industrial sugar cane cultivation area in Colombia, located in the "Valle geografico del Rio Cauca" (216'768 ha). Out of the total area under sugar cane cultivation, 7 plantations (32'215 ha, 15% of the total area) were randomly selected for the LCI data collection. The average LCI values from 7 plantations were used (weighted average by size of the plantation) and further validated as well as complemented by literature data where necessary.

In Colombia, the average crop cycle of sugar cane is 12.6 months and it is cultivated for 6 cycles (ratoon crops) before the plantation is renewed. The average productivity is 112 tons of sugar cane per hectare and year (moisture content 74%). Before harvesting, 70% of the area is burned. The inventory of sugar cane cultivation in Colombia includes the use of mineral and organic fertilizers, pesticides, irrigation, machinery and the field emissions from pre-harvest burning, as well as from the fertilizer and pesticide application. The detailed description of the inventory data is provided in the report "Evaluación del ciclo de vida de la cadena de producción de biocombustibles en Colombia" (Gmünder, S. *et al.* 2012) and will be available in ecoinvent v3.

2.2.4. Oil palm cultivation, Colombia

The inventory for Colombian oil palm cultivation is modelled based on field data from the main oil palm cultivation areas in the North (9'276 ha, representing 8.7% of the total oil palm plantations in Northern Colombia), the Centre (5'850 ha, representing 6.7% of the total oil palm plantations in Central Colombia) and the East (12'445.40 ha, representing 10.3 % of the total oil palm plantations in East Colombia). Overall, the dataset represents 27'571 ha of oil palm cultivation in Colombia out of totally 336'956 ha (8%). The LCI values are validated and if necessary complemented by literature data.

In Colombia, the average life span of oil palms is 25 years, with an average annual productivity of 18.3 tons of fresh fruit bunches per hectare. The inventory includes the use of mineral and organic fertilizers, pesticides, irrigation, machinery and the field emissions from the fertilizer and pesticide application. The detailed description of the inventory data is provided in the report "Evaluación del ciclo de vida de la cadena de producción de biocombustibles en Colombia" (Gmünder, S. *et al.* 2012) and will be available in ecoinvent v3.

2.2.5. Jatropha curcas L. India and Eastern Africa

Jatropha curcas is a non-edible and drought-resistant shrub of the family Euphorbiacae, originating from Central and South America. The shrub, or small tree, is grown throughout the tropics and survives well on marginal soils (Heller, J. 1997). The plant seeds contain 27-40 % high-quality oil (Achten, W.M.J. *et al.* 2008), which is well suited for energetic use (Jongschaap, R.E.E. *et al.* 2007). Additionally Jatropha is cultivated for many purposes such as protection (hedges) (Heller, J. 1997), soil improvement and anti-erosion characteristics (Ogunwole, J.O. *et al.* 2008).

Currently about 900'000 ha land worldwide are under Jatropha cultivation, 80% of which is located in India (GEXSI 2008).

India

The inventory for Indian Jatropha cultivation is based on values derived from agricultural trials. The agricultural test site was established in 2008 at TERI's experimental station at West Godawari, Andhra Pradesh, India. The area is situated at 17° 00' N latitude 81° 10' E longitude with an average daily temperature range of 22°C to 45°C. The annual precipitation is 1098 mm and the plantations are established on red soil with sandy loam texture. Randomized block design (spacing of 3m x 3m) was used for experimental set-up.

Two Jatropha inventories were established: i) a typical small-holder cultivation system with applying little organic fertilizer and water and ii) a typical intensive managed large scale system. In the intensive Jatropha cultivation system, mineral fertilizer was applied at a rate of 200:120:60 g of N:P:K per plant and year. Further, the plantations were drip irrigated, whereas 8 L water per week was applied during 24 weeks. The annual productivity of the intensive Jatropha cultivation system is 2.1 ton per hectare. The extensive Jatropha cultivation system is only irrigated during the establishment of the plantation. Further, three kg of aerobic compost (1.88% available N, 2.56% P and 3.76 % K) were applied per plant only at the time of plantation. The resulting yield is 0.6 ton per hectare, which is relatively low. The detailed description of the inventory data is available in (Gmünder, S. *et al.* in press) and ecoinvent v3.

Eastern Africa

The inventory for East African Jatropha cultivation is modelled based on field data collected in various field visits in 2011. Six Jatropha cultivation areas in Tanzania, Kenya and Ethiopia were visited. The required inventory data was collected by using a structured questionnaire and the gathered field data was validated based on literature values. The dominant cultivation systems in Eastern Africa are small black plantations (about 1 acre) and Jatropha fences. Fences are roughly between 10 and 100 meters long and are typically established as substitutes for deadwood fences. Fences are established for crop and livestock protection and not primary planted to achieve high productivity. Thus, Jatropha fences are not at all managed and all the work is done manually. The annual yield of Jatropha fences ranges from 0.5 kg to 2 kg (average 1kg). Jatropha plantations were mainly established manually, mainly on former agricultural land. Manure was used during the establishment of the plantations to increase the survival rate of the fragile saplings. The annual productivity of Jatropha block plantations in Eastern Africa ranges from 0.5 to 1.5 ton per hectare (average 0.8 ton/ha). The detailed description of the inventory data is available in ecoinvent v3.

2.3. Results

2.3.1. Plausibility checks and validation for nitrate leaching

A full validation with measured values was not possible, since measurements over a wider area are not available. As an alternative we have compared the nitrate leaching values to a Swiss model and to literature values.

The comparison to the model MODIFFUS (Prasuhn, V. *et al.* 2003) for selected crops in Switzerland showed that the values calculated by the model SALCA-NO$_3$ are within the range of the values given by MODIFFUS (Figure 6). The MODIFFUS-value with cereal as preceding crop can be considered as quite close to the expected value, as cereals cover about 2/3 of the arable area in Switzerland. However, nitrate leaching strongly depends on a number of parameters and therefore the real values are likely to deviate from this average value. In all cases the value lay between the minimum and the maximum indicated by MODIFFUS. The outcomes of the model SALCA-NO$_3$ were also validated by experts of ART and found to be realistic.
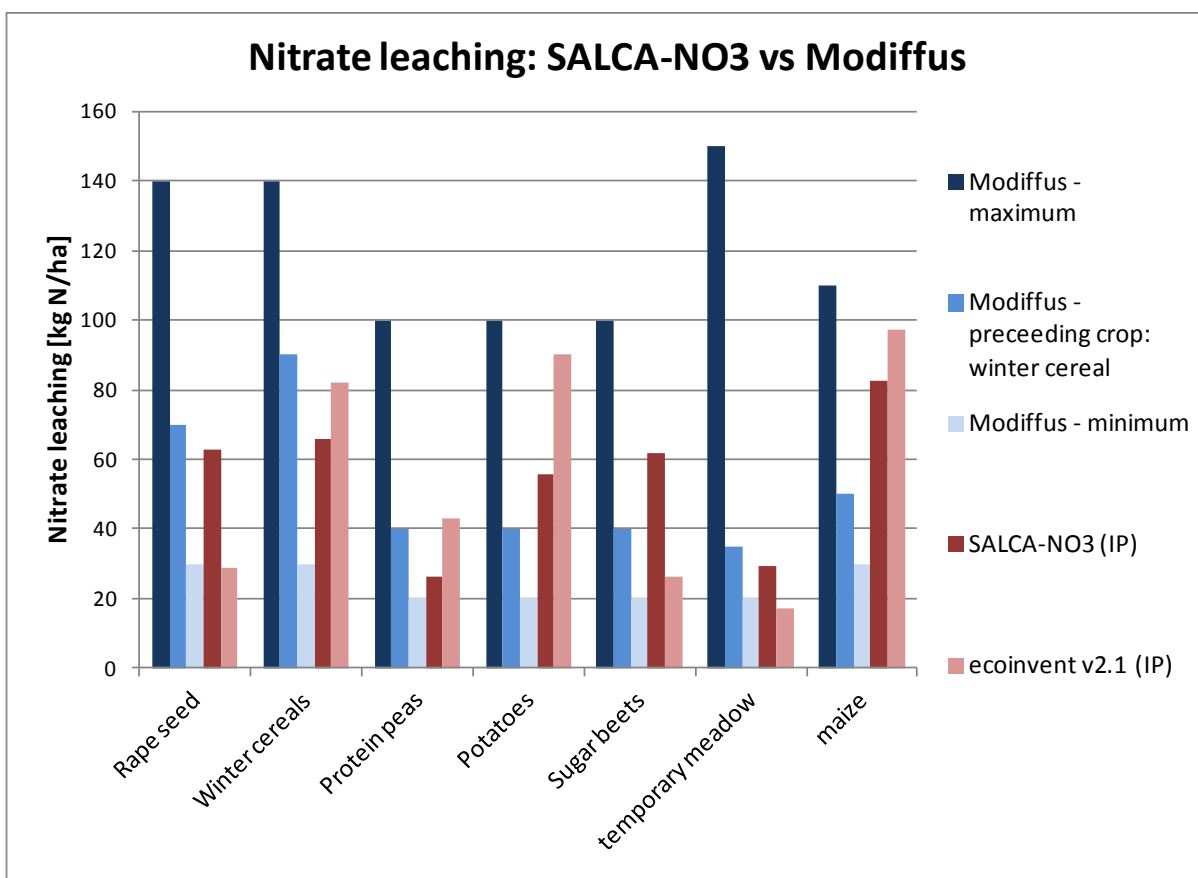
Figure 6: **Comparison of selected nitrate leaching values calculated by the model SALCA-NO$_3$ and by the model MODIFFUS (Prasuhn, V. *et al.* 2003).**

For non-European countries the SQCB-model was used to estimate nitrate leaching. The model results were compared to literature sources. The selection of crops and countries was guided by the available crop production inventories in the ecoinvent database. Nitrate leaching in a given crop depends on a number of factors and therefore a wide range of values can be found, depending on crop management, climate and soil. Figure 7 shows the average values, the mean of the minimum values and the mean of the maximum values given by the different literature sources. No literature values were found for US rice and rapeseed, as well as cotton and sweet sorghum from China. The mean of the minimum values and the maximum values indicated a "central range", which is significantly smaller than the range of the highest and lowest value found in the literature. However, using the latter range would have given an interval that would be far too wide for a reasonable comparison. The figure shows that the values calculated by the SQCB model are either in the range indicated by the literature or do not deviate largely from it. In all cases, where a range of values was indicated in the literature, the SQCB result lay within this range. The estimates can thus be considered as reasonable.
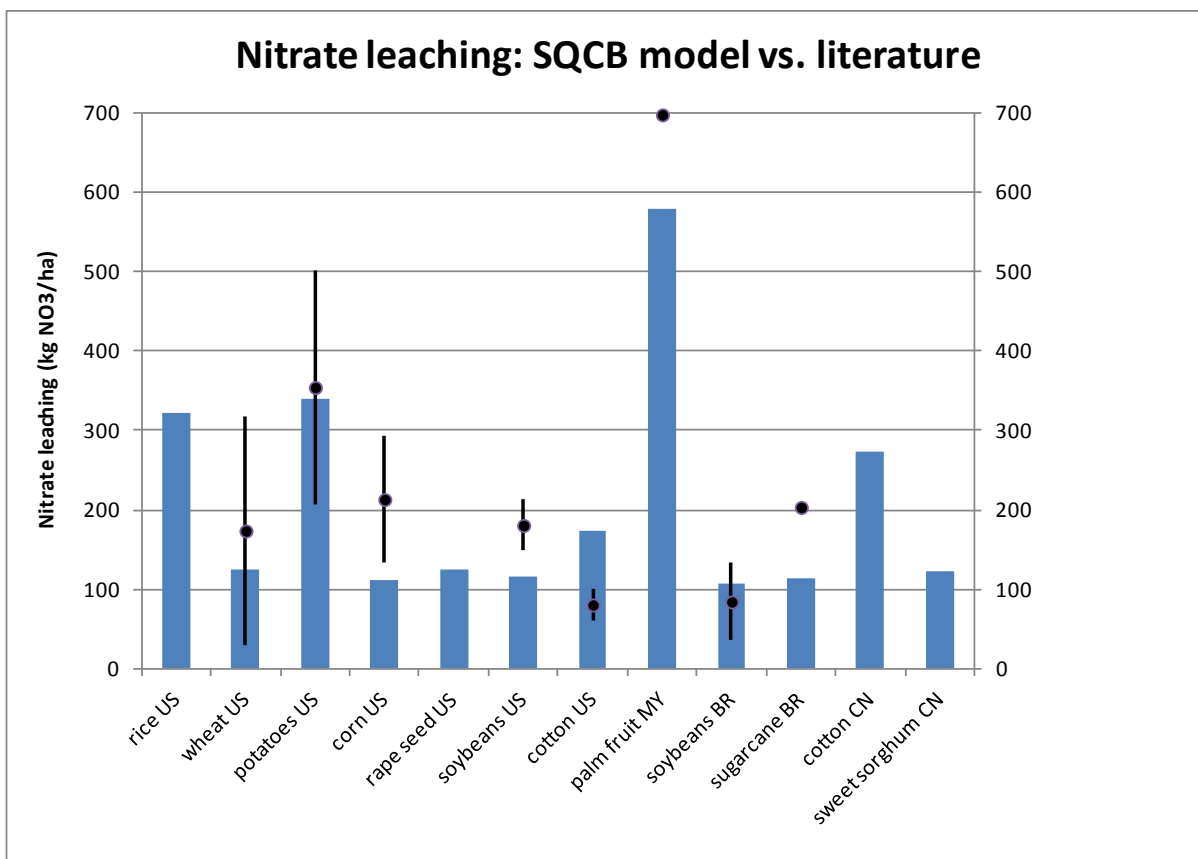
Figure 7: **Comparison of nitrate leaching values calculated by the model SQCB (columns) with literature values: Bullet = mean of literature values, error bar = range from mean of minimum to mean of maximum values.**

2.3.2. Updated nitrogen emission values

The updated ammonia emissions deviate only little from the values in ecoinvent v2.1 (Figure 8 and Figure 9). The biggest changes were found for grass from intensive meadows from integrated production. The changes are explained by different assumptions made in ecoinvent v2.1. The highest ammonia emissions are found in crop inventories where slurry is applied. After the application of slurry, significant amounts of NH_3 are volatilized. The ammonia volatilization after the application of mineral N fertilizers is much lower, except in the case of urea.

In the following graphs a distinction is made between integrated intensive (denoted as IP) and integrated extensive production of cereals and rape seed in Switzerland. The IP production is compliant with the ecological requirements (ökologischer Leistungsnachweis, ÖLN) defined by the Direktzahlungsverordnung. The extensive production is also compliant to the ÖLN, but additionally meets the requirements of the extensive production of cereals and rape seed (the so-called "Extenso" production).
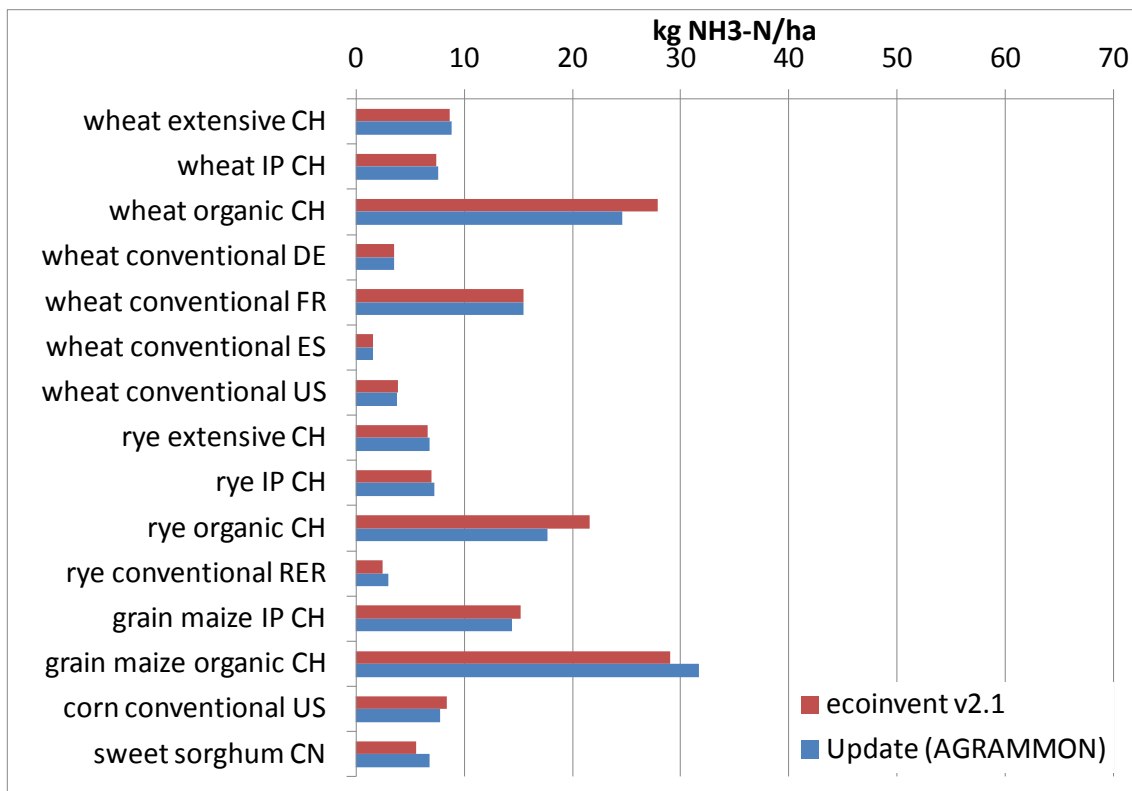
Figure 8: Comparison of the updated ammonia emissions (in kg NH₃-N/ha) for selected cereals to the values in ecoinvent v2.1.
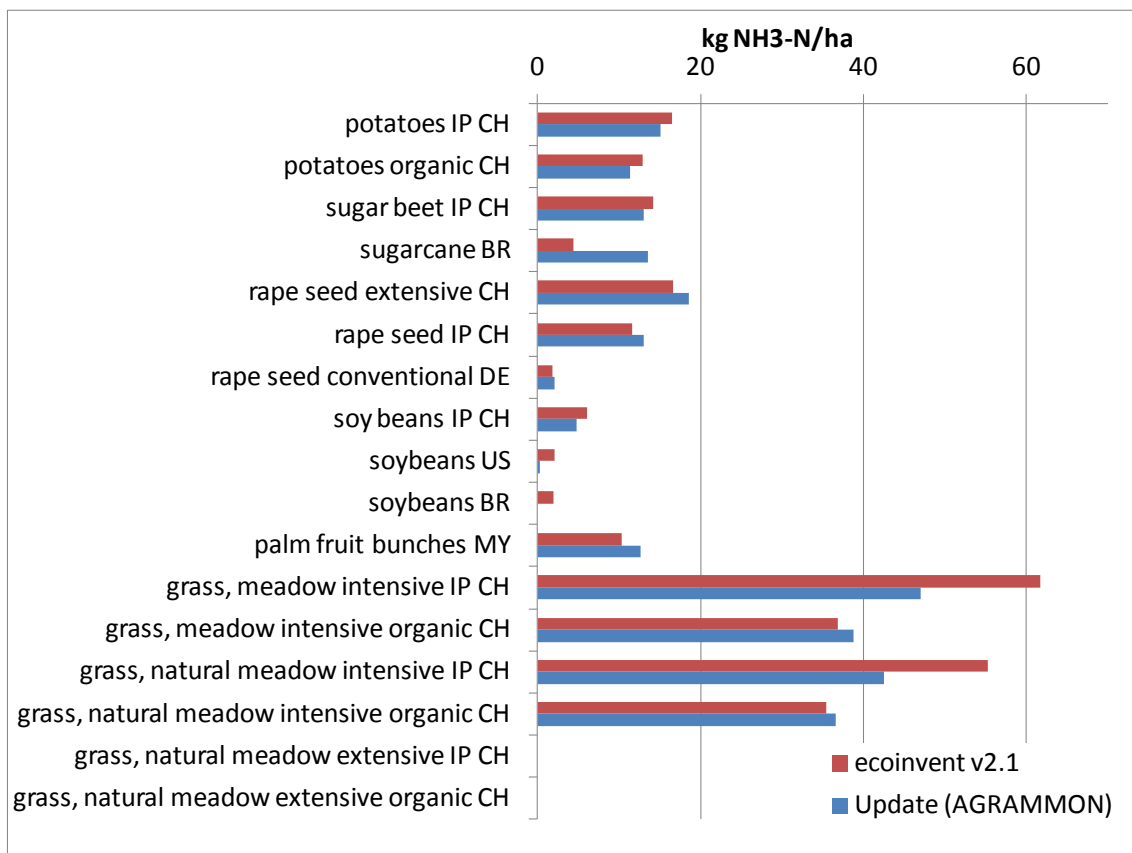
Figure 9: Comparison of the updated ammonia emissions (in kg NH₃-N/ha) for selected crops to the values in ecoinvent v2.1.

The changes in nitrate leaching calculations are much larger than the changes for ammonia emissions (Figure 10 and Figure 11). This has the following reasons: The model SALCA-NO$_3$ has already been applied in ecoinvent v2, but in an earlier version. The updated version is substantially modified, which leads to several important changes in the results. For the non-European crops, different models and emission factors have been applied in ecoinvent v2. This has been harmonized in the new version by consistently applying the SQCB model. For some crops this led to substantial increases (like e.g. for palm oil) or decreases of nitrate leaching (e.g. sweet sorghum in China). As shown by Figure 7, the updated values seem realistic in general (e.g. for palm oil, an even higher value was found in the literature). As an overall tendency, the increases of emissions were more frequent than the decreases (Table 10).
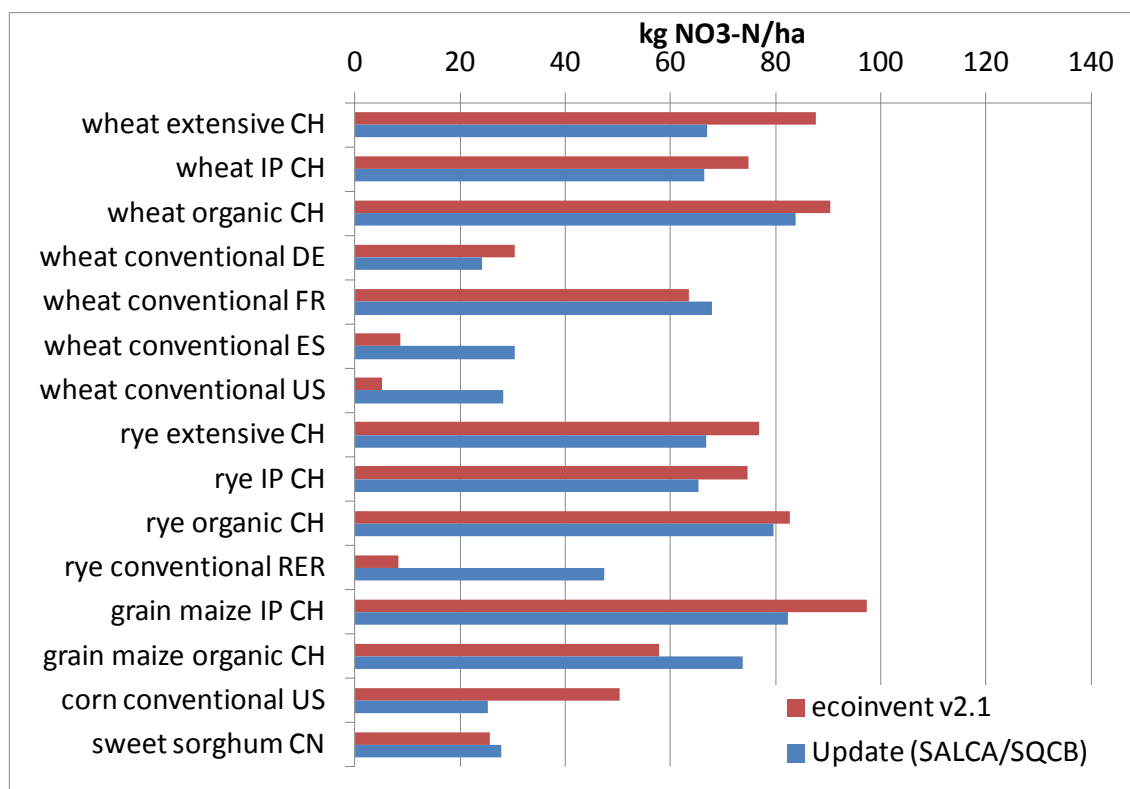
Figure 10:   Comparison of the updated nitrate emissions (in kg NO$_3$-N/ha) for selected cereals to the values in ecoinvent v2.1.
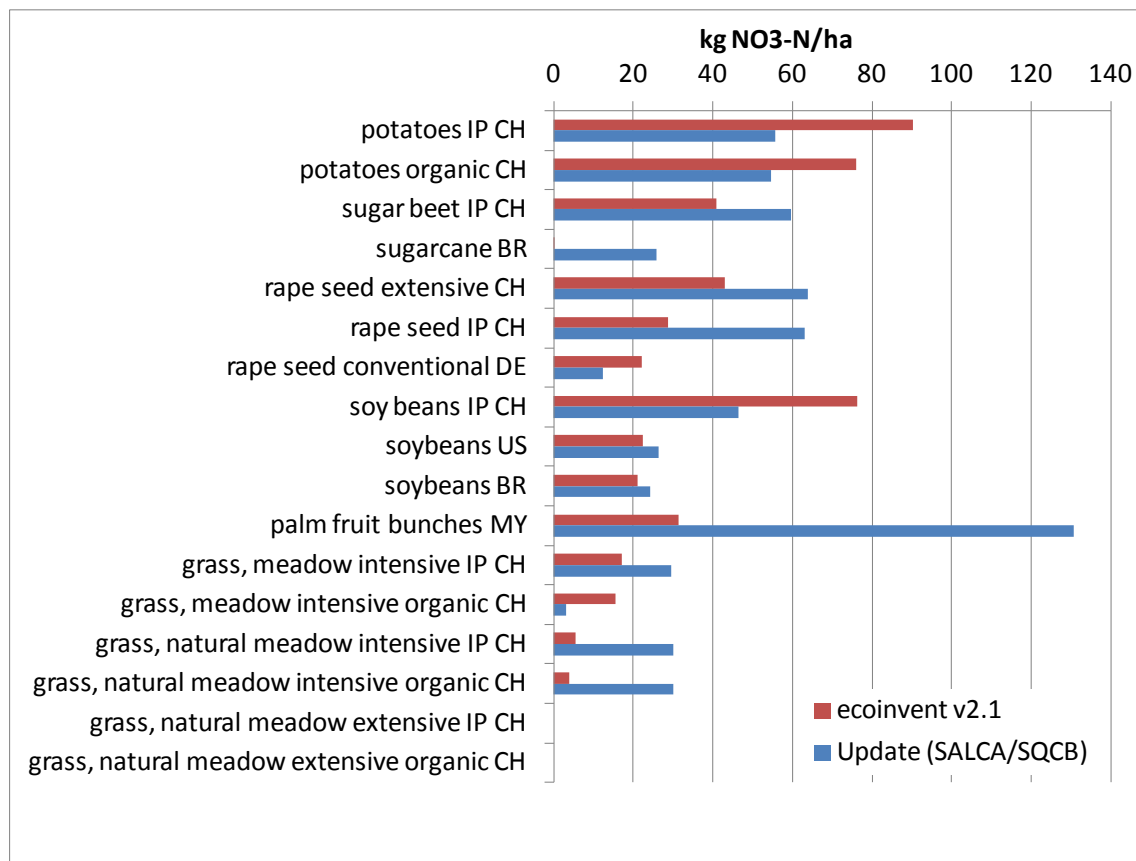
Figure 11: Comparison of the updated nitrate emissions (in kg NO₃-N/ha) for selected crops to the values in ecoinvent v2.1.

The emissions of nitrous oxide show in general a downward trend (Figure 12 and Figure 13). The exact amount of the change depends on several factors:

- Since the effect of symbiotic nitrogen fixation is no more considered as contributing to N₂O emissions according to IPCC (2006), much lower emissions of N₂O for legumes (e.g. peas, beans) can be expected. This can be clearly seen e.g. for the Swiss soybean dataset. The difference for the US and BR soybeans is much smaller than could have been expected, which can be explained by different modelling assumptions made in ecoinvent v2. These modelling assumptions have now been harmonized for v3.

- As a result of the reduced general emission factor for nitrogen fertilization, crop residues, etc. from 1.25% to 1.0%, a general reduction of about 20% can be expected.

- The induced emissions from nitrate leaching have been reduced from 2.5% to 0.75%. This is a very significant reduction of more than a factor 3! In all situations, where nitrate leaching played an important role, this led to a substantial reduction N₂O emissions.

All these effects resulted in reductions of N2O emissions. However, there are also factors that can increase them:

- In ecoinvent v2, the available nitrogen was the basis for the calculation of the N₂O emissions. According to IPCC (2006) the total nitrogen was chosen as the basis for calculation in v3. For mineral fertilizers, this has no effect, since the total amount of nitrogen is assumed to be available to the plants. However, in situations where organic fertilizers are applied, an increase of N₂O emissions can be expected.

- According to IPCC (2006) the emissions of ammonia are no longer subtracted from the total amount of supplied nitrogen. (IPCC 2006) gives the following explanation: „The reason for this change is that field studies that have determined N₂O emission factors for applied N were not adjusted for volatilization when they were estimated."

Depending on the amount of ammonia loss, this can lead to an increase of N_2O emissions.

As both the emissions of NH_3 and NO_3 have been updated, an increase or decrease of the N2O emissions is possible, depending on the amount and the direction of the change of those N emissions.
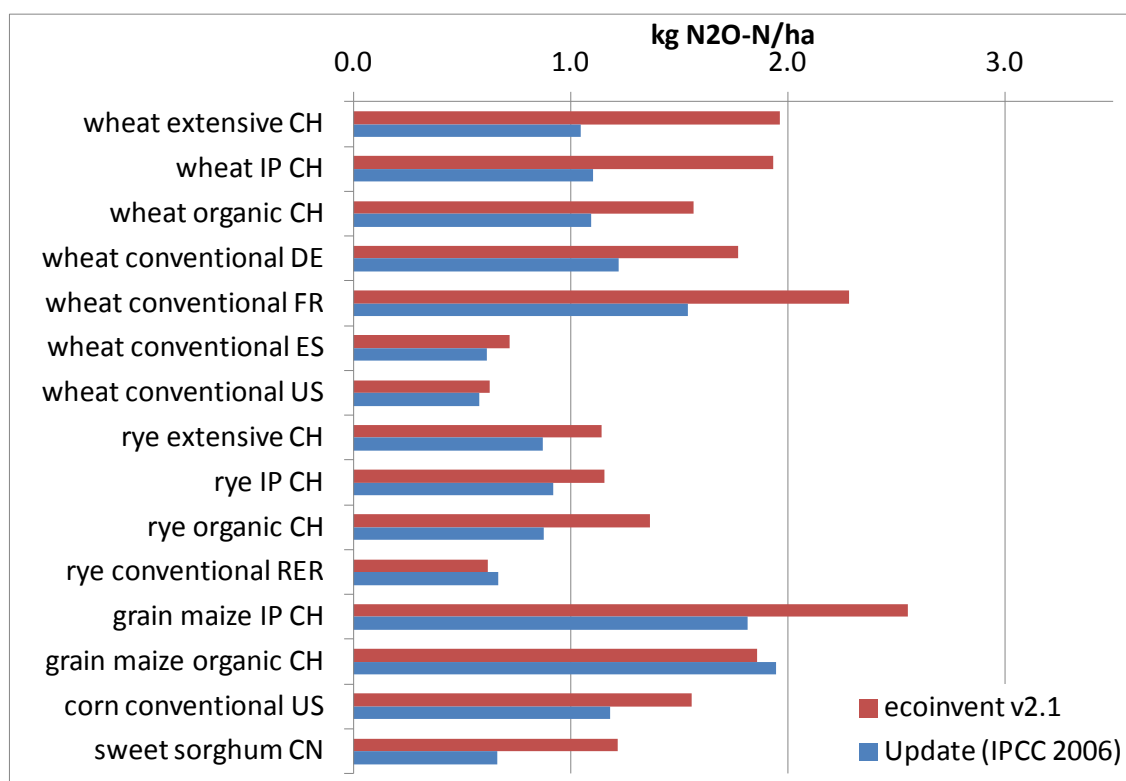
Figure 12: Comparison of the updated nitrous oxide emissions (in kg N_2O-N/ha) for selected cereals to the values in ecoinvent v2.1.
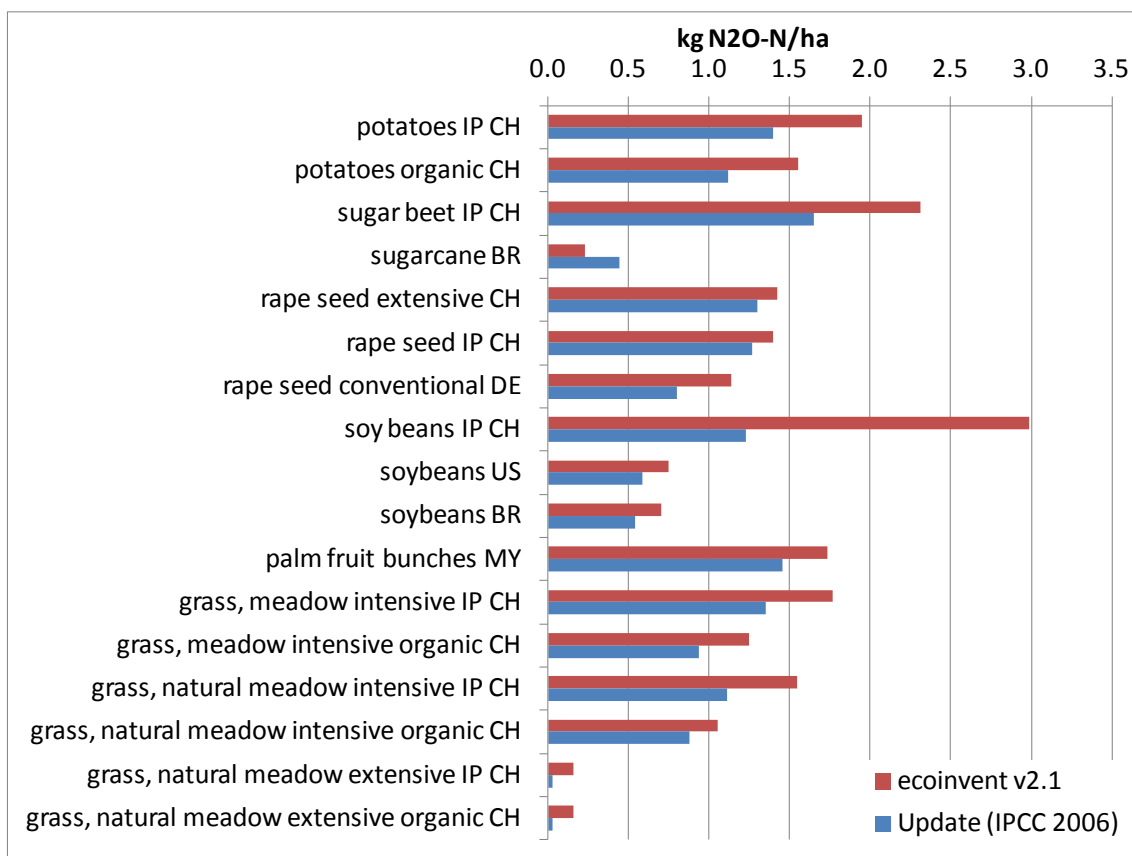
Figure 13: **Comparison of the updated nitrous oxide emissions (in kg N_2O-N/ha) for selected crops to the values in ecoinvent v2.1.**

The overall trend of the relative changes is shown in Table 10. It confirms that the changes for ammonia emissions are generally very small. The average emission decreased slightly. The nitrate emissions increased on average, but this slight trend is the result of important downward and upward trends. The clearest trend has been found for N_2O. The average emissions decreased by more than one quarter. N_2O is a major greenhouse gas of crop cultivation, even more important than CO_2. Therefore, these changes are highly relevant for the assessment of the global warming potential of biomass production.

Table 10: **Summary of the relative changes for the nitrogen emissions for various crop groups.**

	kg NH3-N/ha	kg NO3-N/ha	kg N2O-N/ha
ecoinvent v2.1	13.69	40.28	1.38
ecoinvent v3	13.04	45.70	1.01
Relative change	-4.8%	+13.4%	-26.4%

2.3.4. Updated LUC emission values

Figure 14 shows the updated LUC emissions for the updated (SB, FFB and SC) and new crop activities per ha crop cultivated. The updated crop activities are shown in comparison to ecoinvent v2.2.
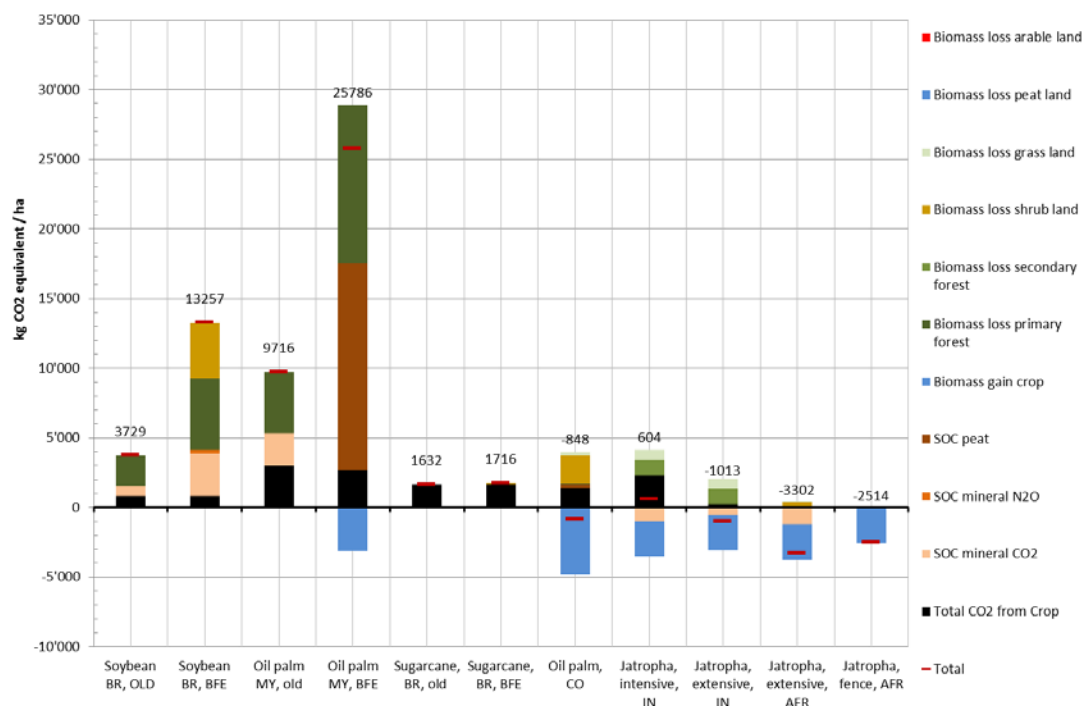
Figure 14: Comparison of the updated LUC emissions (in kg CO₂equiv./ha⁻¹) for selected crops to the values in ecoinvent v2.2.

Comparing between ecoinvent v2.2 and ecoinvent v.3.0, the global warming potential (GWP) of soybean cultivation in Brazil increases by the factor 3.5 while the cultivation of oil palms in Malaysia increases by the factor 2.7. The main reason for this increase is the consistent consideration of the full amount of C stored in AGB, BGB, DOM and SOC. That is, even though the land transformation per ha crop cultivated decreased for both crops (see Table 1), the consideration of all relevant carbon pools dominates the outcome.

The oil palm plantation in Colombia does show a net increase of the carbon stock, since the carbon stock of the tree plantation is generally higher than the carbon stock of the former land use. The same is true for the jatropha cultivation systems, which are established on rather low carbon stock land (mainly agricultural land, see Figure 14). Even though the cultivation of a tree crop on agricultural land generally does lead to a direct carbon stock increase, it has to be kept in mind that replacing agricultural crops might cause indirect land use changes, which potentially increases indirectly the pressure on natural ecosystems (the potential effect of indirect land use shifts are discussed in section 4.7.5).

Figure 15 shows the updated LUC emissions for the updated (SB, FFB and SC) and the new crop activities per kg crop cultivated.
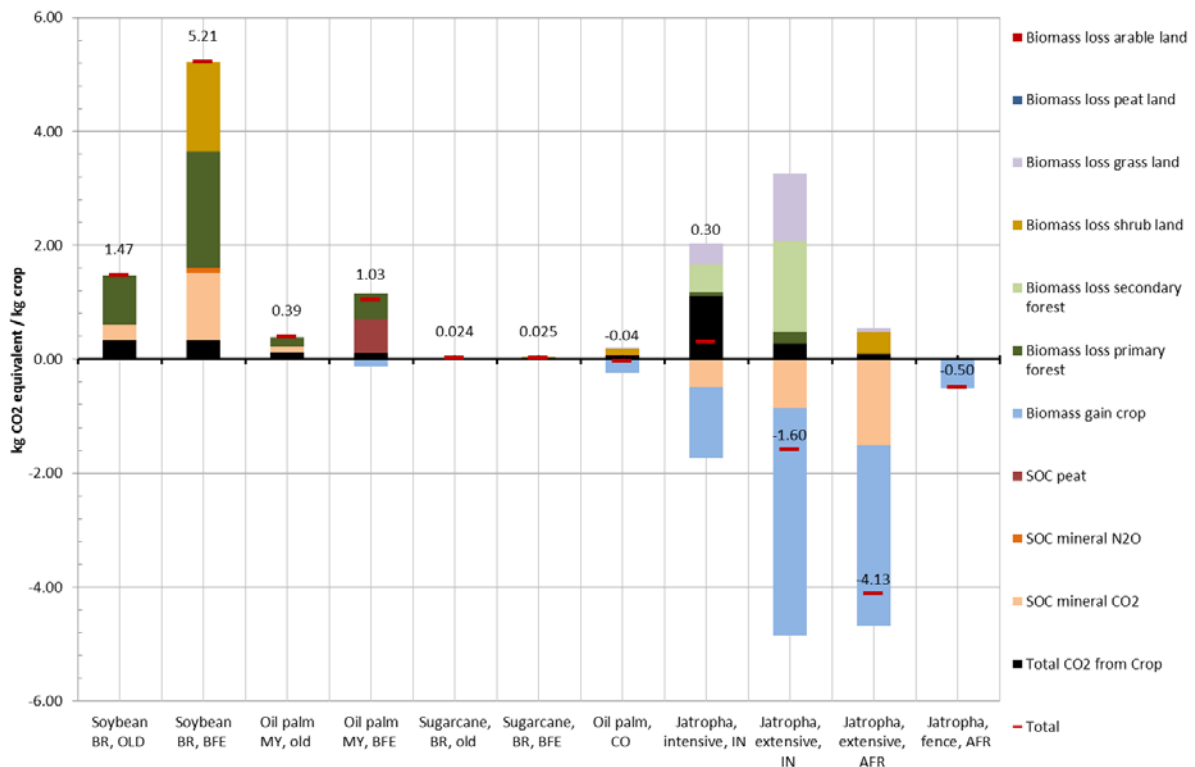
Figure 15: Comparison of the updated LUC emissions (in kg CO$_2$equiv./kg crop) for selected crops to the values in ecoinvent v2.2.

Regarding the cultivation of soybean in Brazil, 6% of the GWP are caused by the crop cultivation, while roughly 25% of the emissions are related to SOC loss caused during the land occupation. The remaining 69% are related to the transformation of primary forest (39%) and shrublands (30%).

The results of oil palm cultivation in Malaysia are dominated by the occupation impacts associated with peat oxidation (58%) and the transformation of primary forest (44%). The permanent gain in AGB/BGB causes a benefit of -12% that overcompensates the impacts resulting from crop cultivation (10%).

Even though the carbon stock changes related to jatropha cultivation are relatively small per hectare, the carbon benefit per kilogram seeds harvested is substantial. The reason for this is that the yields, ranging from 0.6 to 2.5 t seeds per hectare, are relatively low. Consequently, the increase in carbon stock due to LUC is allocated only to a little amount of seeds, leading to a relatively high carbon benefit per kg seed.

3. Conversion processes

3.1. Introduction

This module looked at the preparation and conversion of raw materials and low quality energy carriers into higher quality energy carriers. The final process for all substances was combustion in centralised or decentralised plants in order to generate electricity or electricity and heat, or as fuels in road passenger vehicles. Table 11 shows the range of primary energy carriers together with their conversion and combustion processes which have been analysed in this module.

Table 11: Conversion and combustion processes analysed in Module 2 (excluding municipal waste incineration processes, which are described later in this section).

Primary energy carrier	Conversion process 1	Conversion process 2	Conversion process 3	Energy carrier	Combustion technology
SRC[1] Salix	Harvesting and chipping	Drying	None	Biomass chips	Thermal power plant for CHP[5]
Annually harvested Miscanthus	Harvesting and chipping	Drying	None	Biomass chips	Thermal power plant for CHP
Alfalfa grass	Harvesting	Anaerobic digestion	None	Biogas	ICE[6] CHP plants
			Purification	SNG[4]	ICE[6] CHP plants
Animal slurry	Hydrothermal gasification	None	None	SNG[4]	ICE[6] CHP plants
					CC[7] power plant
					ICE[6] Euro3[8] Passenger car
Wet wood	Hydrothermal gasification	None	None	SNG[4]	ICE[6] CHP plants
					CC power plant
					ICE[6] Euro3[8] Passenger car
Sewage	Anaerobic digestion to biogas	BCM[2] process [2]	None	SNG[4]	ICE[6] CHP plants
					CC power plant
					ICE[6] Euro3[8] Passenger car
Jatropha seeds	Harvesting and drying	Oil extraction & purification	Transester-ification	Biodiesel	ICE[6] Euro3[8] Passenger car
Canadian oil sands	Bitumen extraction	Upgrading to SCO [3]	Refining	Petrol & diesel	ICE[6] Euro3[8] Passenger cars

[1] Short rotation coppice

[2] Biogass-CO_2-Methane

[3] Synthetic crude oil

[4] Synthetic natural gas

[5] Combined heat and power

[6] Internal combustion engine

[7] Combined cycle (Gas and steam turbines)

[8] Emissions standard compulsory for all new cars sold between 2000 and 2005. Although this standard now represents a somewhat aged technology it was used in this study in order to allow an LCA-based comparison of the biofuels presented here with those from the previous Swiss biofuels study (Zah, R. *et al.* 2007).

3.1.1. Solid energy carriers

The solid biomasses of salix and miscanthus are perennial plants conventionally grown on large-scale purely as energy crops. Salix is harvested on a short rotation coppice basis approximately every three years whereas miscanthus is harvested annually. Both energy crops can be harvested as stems or chipped directly during the harvest. Salix has typical moisture content on harvesting of around 55% and so harvesting in stem form can enable them to be dried naturally in the air if simply stacked under cover, whereas salix chips require mechanically induced drying (Dawson, M. 2011). Being a grass, miscanthus on the other hand has a typical moisture content at harvest of less than 30% and in dry storage will dry down to 15% or less without forced drying measures (Caslin, B. *et al.* 2011). Miscanthus is fired in shredded form but can be supplied in baled or in pre-shredded form and therefore harvesting can be done to suit the storage preferences. Both biomass fuels can be fired singularly – usually in small scale CHP or co-fired in centralised thermal power plants together with conventional feedstocks such as coal or lignite. With relevance to the Swiss situation this study considers salix or miscanthus as unmixed and individual feedstocks combusted in small scale CHP (<10MWth) and suitable to a very local demand of the heat energy.

3.1.2. Liquid energy carriers

The biomass sourced liquid fuels considered were that of biodiesel from the transesterification of seed oil and ethanol from the fermentation of various crops. The crop is grown processed to oil and biodiesel outside of Switzerland. The ethanol and biodiesel are then available for use in a wide variety of ICE applications but for the purposes of this study and to provide a basis for comparison we considered their use in a passenger car.

The project goals also encompassed an alternative source of liquid fossil fuels for use in road transport which focused on the growing production of SCO from Canadian oil sands. This was done in order to highlight the sensitivity of the LCA results for fossil resources. The mining and processing of oil sands is well known as an unconventional route to liquid fossil fuels. As such it presents a source for comparison with crude oil extracted by more conventional means and in parts of the world which are currently the major suppliers to the Swiss market. In this regard it must be made clear that crude oil from Canadian oil sands (otherwise known as synthetic crude oil – SCO) is of no relevance to the Swiss supply mix – either currently or foreseen for the future due to the strength of the North American market. However, the mining and upgrading of oil sands is an established, developed and regulated industry which represents a very different source of crude oil and for which data availability is relatively high. Other unconventional fossil energy resources of potential future significance to the Swiss supply mix were regarded as being either insufficiently unconventional (e.g. deep-sea deposits) or insufficiently developed (e.g. shale gas). Budget limitations also lead to the limitation to one source and further research is therefore required in order to evaluate the potential environmental impacts of future fossil fuel supply to Switzerland. Within such further work should also be an update of the conventional fossil fuel production pathways.

3.1.3. Gaseous energy carriers

For the gaseous energy carriers the focus lay with those of a quality and composition similar to natural gas and therefore suitable to allow feeding into the natural gas pipeline network. The analysis considered a conventional route using an agricultural crop feedstock followed by anaerobic digestion to produce biogas and then a pressure-swing adsorption (PSA) purification process to achieve an SNG. Analyses of the combustion of the biogas and the SNG in ICE CHP plants were considered in order to determine potential benefits to be gained from purifying the biogas prior to combustion.

Hydrothermal gasification as a single process to convert raw biogenic feedstocks directly to SNG is currently under development as a potentially viable route to utilise biomass with high water content without the need for to dry it. The process operates under conditions which are supercritical for water – meaning high temperature and pressure – such that the water itself acts as a catalyst to methanise the biomass whilst the salts fall out of solution and are

collected separately. This technology is currently undergoing advancement at the PSI in order to test it at the pilot scale. The raw biomass feedstocks considered here are that of cattle slurry and wet wood direct from forestry operations.

The BCM process uses a highly effective amine washing of biogas in order to purify the methane content and to remove harmful trace elements. In the scenario analysed in this study the biogas feedstock is from a sewage treatment facility.

SNG's from hydrothermal gasification and from the BCM process are considered suitable as pure fuels for decentralised CHP plants and for use in road passenger transport. It is unlikely that SNGs from these sources would be used in pure form in centralised combined-cycle (CC) power plants as this would either involve a transport of the primary fuels to the processing plant located in proximity to the CC plant, or a distribution network for the de-centrally produced SNGs for transport to the CC plant – both of which would incur inacceptable cost and effort. A more feasible scenario would be to inject the SNGs into the natural gas network at the source of production and to thereby dilute the fossil natural gas in the network with biogenic SNG. However, whilst the modelling of the production on a life-cycle basis would be feasible, modelling the combustion would become irrelevant as the ratio of SNG to natural gas at the CC plant would become very small. For the purposes of gaining insights into the effect of SNG combusted in CC plants we therefore analysed the hypothetical scenario of SNG produced de-centrally and then used in pure form in a CC plant.

3.1.4. Passenger vehicle operation

For the operation of passenger vehicles in Europe the study used new datasets representing updated, corrected and extended versions of those available in ecoinvent v2.2 (Spielmann, M. *et al.* 2007). Emission factors were determined using the same institutional sources as for v2.2 but with updated versions of the data made available. The Tremove database (T&M-Leuven 2007) provided emission factors for CO_2, CO, NO_x, CH_4, exhaust particulates (PM) and total VOC. All other emissions including NMVOC splits, additional substances as well as all non-exhaust emission factors were taken from more recent publications of the European Environment Agency (Ntziachristos, L. *et al.* 2009a; Ntziachristos, L. *et al.* 2009b).

The updates and corrections were done as part of a more general work to expand the vehicle options available in ecoinvent. A key objective was therefore to generate emission factors which followed logical and consistent trends when considered in the broader context of differing vehicle sizes and emissions standards. The aggregated differences between the v2.2 and v3 datasets for operation of a mid-size petrol fuelled passenger car of Euro3 standard are given in Figure 16. The more significant differences seen in the graph are therefore the result of this broader consistency and the corrections (particularly N_2O, $PM_{2.5}$ and CH_4) and the increase in emissions profile (particularly PAHs, heavy metals and emissions to water and soil).

The technology standard of the study remains to be Euro3 (the emissions standard for all new cars sold between 2000 and 2005) for the purposes of comparison with the 2007 study (Zah, R. *et al.* 2007). Clearly results would be somewhat lower if the present emissions standard (Euro5: 2009 to 2014) were taken into account but such updates were not within the aims of the project. It was instead intended to generate a consistent and level basis for comparison across fuel options.
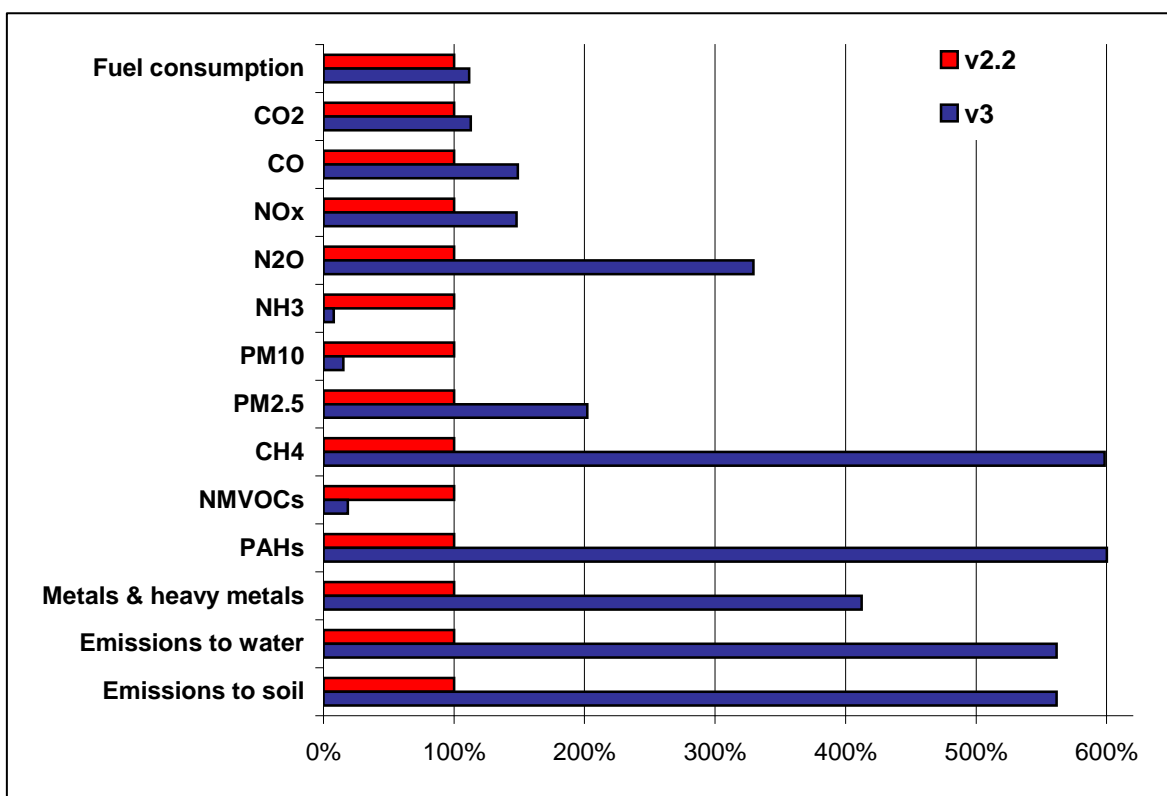
Figure 16: Fuel consumption and emissions for a mid-size petrol-fuelled passenger car of Euro3 standard as given in ecoinvent v2.2 (Spielmann, M. *et al.* 2007) and the differences of new operation datasets intended to be added to ecoinvent v3.

The operation of passenger vehicles using the alternative fuels here under analysis was done based on the principal of equal energy demands meaning that, for example, the consumption of biodiesel is equal in energy terms to the consumption of diesel in a diesel engine. These aspects are quantitatively given in Table 12.

Table 12: Fuel consumptions of mid-sized Euro3 passenger vehicles operated on the fuels relevant to the present study.

	Unit	Petrol	Ethanol	Diesel	Bio-diesel (Jatropha)	Bio-diesel (Others)	Natural gas	SNG (Slurry)	SNG (Wood)	SNG (BCM)
Energy density (LHV)	MJ/kg	42.9	26.7	42.8	39	37.6	48.6	51.5	49.2	48.1
Energy demand	MJ/km	2.86	2.86	1.82	1.82	1.82	2.51	2.51	2.51	2.51
Fuel consumption	kg/km	0.0674	0.107	0.0425	0.0467	0.0484	0.052	0.0487	0.051	0.0522

3.2. Processing and conversion of solid fuels

3.2.1. Salix from agricultural short rotation coppice (SRC) production

The conversion of salix in Module 2 begins with the harvested product in chipped form and with moisture content (MC) of 55%. The storage of salix woodchips at this level of MC would rapidly lead to fungal decay and it so it is therefore necessary to reduce this down to approximately 30% (Dawson, M. 2011). Drying of the pile is done using electric-powered ventilation of ambient air and which requires an electricity use of 10 kWh/m^3 (Helin, M. 2005). It is assumed that there is no waste heat from the CHP plant which could assist this process. Drying is assumed to take place at facilities on the site of the CHP plant meaning that the transport of the woodchips uses the higher density at 55%MC. For transport distances

relevant to a relatively dispersed production in Switzerland we adopt the average distance of 63km used in ecoinvent v2.2 for the transport of mixed woodchips as a waste product from industry.

3.2.2. Miscanthus from annual agricultural production

The conversion of miscanthus in Module 2 begins with the harvested product in shredded form and with a moisture content (MC) after harvest and air drying of 15% (Caslin, B. *et al.* 2011), meaning that further drying is not necessary in order to avoid further decay if simply dry stored. Similarly to salix, the production of miscanthus is not assumed to be concentrated in very close proximity to the CHP plant and so the average transport is also assumed to be 63km although the higher energy density but relatively low volumetric density of the fuel leads to around 13% less tkm per MJ combusted.

Table 13: **Combustion-relevant compositions of salix and miscanthus in comparison with the reference product of mixed woodchips from industry from the ecoinvent database v2.2. Compositions are averaged data from (Reisinger, K. *et al.* 1996; Van-Loo, S. *et al.* 2008) for Salix and ((Reisinger, K. *et al.* 1996; Hersener, J.-L. *et al.* 1997; Scurlock, J.M.O. 1999; Newman, R. 2003; Dahl, J. *et al.* 2004; Friedl, A. *et al.* 2005; Van-Loo, S. *et al.* 2008) for miscanthus.**

	Unit	Ecoinvent	This study	
		Mixed wood-chips	Salix woodchips	Miscanthus, chopped
Moisture content (MC) at combustion	%	40	30	15
Density (@ 0% MC)	kg/m^3	189	240	74
Density (wet basis)	kg/m^3	264 @40% MC	312 @30% MC	85 @15% MC
HHV	MJ/kg	20.2	19.2	19.2
LHV	MJ/kg	13.2	13.2	15.3
LHV	MJ/m^3	3463	4124	1299
Carbon	% dry matter	49.4	48.3	47.9
Hydrogen	% dry matter	6.10	6.07	5.4
Nitrogen	% dry matter	0.25	0.44	0.4
Sulphur	% dry matter	0.01	0.04	0.1
Chloride	% dry matter	0.001	0.014	0.2
Oxygen (by difference)	% dry matter	44.1	43.6	42.5
Ash	% dry matter	1.0%	0.8%	3.2%

3.2.3. Solid fuel combustion

The combustion characteristics of salix and miscanthus with relevance to the emissions regulations in Switzerland was analysed during a previous study for the BfE (Hersener, J.-L. *et al.* 1997) and for a technical account of the relevant factors we therefore refer to this work. The use of salix woodchips does not pose any significant differences in combustion characteristics when compared to forest-sourced woodchips. The difference in combustion characteristics between forest-sourced wood chips and salix (field-sourced) wood chips is not significant as long as the salix contains a majority of woody biomass from several years of growth and that it can be converted into woodchips of similar dimension to that of forest-sourced woodchips.

Miscanthus, however, is harvested every year and as a form of grass it does not build up the same woody mass as does salix. When biomass consists of a high proportion of younger, non-woody material then the content of ash, potassium, chlorine and other elements will tend to be significantly higher. With regard to combustion, the thermal characteristics of chopped miscanthus are only insignificantly different to that of wood chips such that in reality the physical form of the fuel is far more influential than the thermal properties of the organic

substances. As with salix therefore, we assume that on entering the combustion chamber the miscanthus chips will be of a homogenous nature and similar in size to the woodchips and that these can be evenly distributed within the combustion chamber. Although the same basic technology can be used for combustion in CHP plants, the higher concentration of ash and its lower melting point can cause slagging on the grate and in the combustion chamber. The particular composition of miscanthus will also lead to the formation of salt deposits (KCl, K_2SO_4) and thereby to increased corrosion. The volatile elements will vaporize during combustion and then condense to fine particulates in the flue gas (Hersener, J.-L. *et al.* 1997; Dahl, J. *et al.* 2004). In Table 13, the content of nitrogen in forest-sourced wood is reported to be 0.25wt% whereas (Hersener, J.-L. *et al.* 1997) determine this to be as little as 0.08wt% and for miscanthus as high as 0.58wt% - thus giving a much larger difference to that shown here in Table 13. Nitrogen content does however vary significantly; for wood this can be between 0.05-0.65wt%, for miscanthus 0.06-0.87wt% (Hersener, J.-L. *et al.* 1997) but, in general, higher nitrogen content implies higher NO_x emissions. The P and K content influence the nutrient content of the residue. Heavy metals determine the toxic level of the residue as well as that in the exhaust gas.

In order to operate a miscanthus-fuelled CHP plant a number of measures will therefore need to be taken in order to limit the slagging and build-up of corrosive deposits and to keep emissions within permitted limits. The formation and deposition of salts in the combustion system can be eased with the use of water cooled combustion chamber walls, efficient grate cleaning, mechanisms for cleaning the boiler, etc. The melting point of wood ash (<1100°C) is significantly higher than that of miscanthus ash (>850°C) and so the combustion should be kept at a lower temperature than in wood firing systems. The lower combustion temperature then has an influence on the type of de-NO_x flue gas treatment which is needed. The content of N, S, Cl & K have a relative influence on the emissions and flue-gas cleaning required. They are significantly higher than for wood. Due to the high proportion of minerals in miscanthus, PM emissions after cyclone filter of between 300mg/Nm^3 and 100mg/Nm^3 are still to be expected, meaning that additional filtering using particle separators will be needed (electric, mesh, ceramic). If a fabric filter is used then it will be necessary to use a lime additive in order to enable HCl adsorption. Grate and cyclone ash are suited to use as waste fertilizer but filter ash must be treated like that from KVA and be disposed of.

Table 14: **Specific changes to the woodchip combustion dataset for salix and miscanthus fuel.**

Fuels	Changes made
Salix wood chips and chopped miscanthus	Demand based on energy density and equal efficiency as for mixed woodchips.
Materials	**Changes made**
Lubricating oil	For salix increased by 20%, for miscanthus by 40%
Chemicals for flue gas cleaning	For salix increased by 20%, for miscanthus by 40%
Road transport	Based on wet basis mass per MJ and same transport distance as mixed woodchips
Cogen unit ORC: components for electricity and common components for heat & electricity.	Increased to account for significant infrastructure additions in combustion and flue gas unit. Salix 20%, Miscanthus 50%.
Cogen unit ORC building	According to energy density and therefore the storage space needed.
Emissions to air	**Changes made**
Carbon dioxide, biogenic	Based on the carbon content of fuel
Nitrogen oxides	Based on data from Reisinger 2009. Own calculations made using N content of fuel, NO/NO2 ratios. Verified by the similarity between calculated and existing values for mixed woodchips NOx (can be interpreted as a technology specific Nox factor.
Particulates, < 2.5 um	Based on Reisinger 2009.
Sulfur dioxide	Based on S content of fuel and content of S going to flue gas (Van-Loo 2008).
Waste to treatment	**Changes made**
Disposal of wood ash mixtures to municipal incineration, agricultural use and sanitary landfill.	Based on the ash content of the fuels

3.3. Production and use of liquid fuels

3.3.1. Jatropha oil and biodiesel production

After harvesting of the jatropha seeds, the remaining seed husks are removed and brought back on the field. The jatropha seeds are transported from the field to the oil extraction factory over a distance of 50km by a 16t lorry. The oil is extracted by cold pressing using an electric screw press and purified by a bag filtration system (Gmünder, S. *et al.* in press). The press has an oil expelling efficiency of 80% and the filtration system has an efficiency of 92%. The oil content of the jatropha seeds is 35%, resulting in a seed demand of 3.88 kg per kg purified oil. The expeller has a capacity of 175 kg seeds per hour and is powered by a 22 kW generator. The lifespan of the press is assumed to be 10 years, while it is operated 24 hours a day. Per kg seeds also 2.88 kg of press cake is produced.

A base catalytic transesterification reaction [44] was modelled to produce jatropha methyl ester (JME) out of J. curcas oil and methanol (134 kg per ton JME). The reaction wsas carried out in a continuous flow reactor with a capacity of 1000 l/h. The catalytic reaction was mediated by potassium hydroxide. A mass conversion efficiency from J. curcas oil to methyl

ester of 95% is assumed. Besides JME glycerine is also produced as a by-product. A mass fraction of 0.09 of glycerine in relation to JME was used and the glycerine is substituted for conventional glycerine.

As shown in Table 11, module 2 begins with biodiesel from jatropha in its final fuel form. For the determination of combustion characteristics it was firstly necessary to define the composition and properties of the biodiesel in relation to those of the low sulphur diesel reference fuel existing in ecoinvent (Jungbluth, N. 2007). The compositions and properties of biodiesel were averaged from (Hanumantha Rao, Y.V. *et al.* 2008; Huang, J. *et al.* 2010; Jindal, S. *et al.* 2010; Jingura, R.M. *et al.* 2010; Ganapathya, T. *et al.* 2011) and are also shown in **Table 15**. Petrol and diesel from oil sands have the same composition as those from conventional crude oils. Biodiesel, however, has significant differences which effect combustion and emissions.

Table 15: **Compositions of the liquid fuels (Jungbluth, N. 2007).**

	Unit	Petrol (low S)	Diesel (low S)	Petrol (low S)	Diesel (low S)	Biodiesel
		(ecoinvent)	(ecoinvent)	Oil sands	Oil sands	Jatropha
Carbon	Wt%	86.5	86.5	**86.5**	**86.5**	**77.1**
Hydrogen	Wt%	13.5	13.3	**13.5**	**13.3**	**11.8**
Oxygen	Wt%	0.3	0	**0.3**	**0**	**11.0**
Nitrogen	Wt%	0	0	**0**	**0**	**0**
Sulphur	Wt%	0.005	0.005	**0.005**	**0.005**	**0.002**
Heavy metals	mg/kg	2.85	2.85	**2.85**	**2.85**	**0**
LHV	MJ/kg	42.5	42.8	**42.5**	**42.8**	**39.0**
HHV	MJ/kg	45.1	45.4	**45.1**	**45.4**	**40.3**
Density	kg/l	0.75	0.84	**0.75**	**0.84**	**0.9**
Cetane number			46*			**53.9**
Viscosity @40°C	cSt		2.5*			**4.5**

* Values taken from (Kinast, J.A. 2003).

3.3.2. Combustion of jatropha biodiesel in road transport vehicles

Since the biodiesel can be used simply as a substitute for conventional diesel, its use in road transport vehicles was considered. Operation datasets were compiled using the diesel fuelled processes as reference data (for vehicle related datasets see Figure 16). Biodiesel fuel consumption was calculated based on its energy density in relation to that of diesel (see **Table 15**). CO_2 and SO_2 emissions were calculated according to the carbon and sulphur content of the biodiesel. CH_4 as a fraction of VOCs used the same relationship as that for conventional diesel. N_2O and NH_3 emissions were assumed to be the same as for diesel due to no information being found to the contrary. Other emissions were calculated as fractions of diesel emissions according to (Graboski, M.S. *et al.* 2003; NBB 2012) and as shown in Table 16.

Table 16: **Emissions from biodiesel as percentages of those from conventional diesel com-bustion in internal combustion engines (Graboski, M.S. *et al.* 2003; NBB 2012).**

	CO	NOx	Total HC	PM	VOF as % of PM	PAH	Formal-dehyde	Acetal-dehyde
Diesel	100%	100%	100%	100%	100%	100%	100%	100%
Jatropha biodiesel	55.2%	108.7%	54.6%	41.9%	22.2%	20.0%	79.9%	69.5%

3.3.3. Ethanol combustion in passenger vehicles

For purposes of achieving consistency with the use of the other transport fuels and to aid comparison with the results from (Zah, R. *et al.* 2007), the emissions from the use of ethanol in passenger cars were re-modelled. As (Brown, G. 2008) describes in their review of combustion research, determining representative emissions factors for ethanol use has not yet been consistent and conclusive, and may well lead to decreases in certain emissions but increases in others. As shown in **Table 17**, the composition and thus also the energy content of ethanol differ significantly to that of petrol, therefore leading to significant differences in the emissions resulting from combustion.

Table 17: Properties and composition of ethanol in comparison to that of petrol, as given in ecoinvent (Jungbluth, N. 2007) and in (Brown, G. 2008).

	Ecoinvent	BEST		Unit
	Petrol	Petrol	Ethanol	
Carbon	86.5	86.5	52.2	wt%
Hydrogen	13.5	13.5	13.1	wt%
Oxygen	0.3	0	34.7	wt%
Nitrogen				wt%
Sulphur	0.005			wt%
Density	0.75	0.75	0.794	kg/l
LHV	42.5	42.9	26.7	MJ/kg
Octane n		90	100	
Cetane n		12.5	8	

The addition of ethanol to petrol as a blend is broadly understood to result in the reduction of a number of the common pollutants, and unlike petrol or diesel, ethanol does not contain alkenes (forms of VOCs), aromatics (PAHs) or sulphur. The relatively high oxygen content of ethanol aids a more complete combustion of the fuel which leads to lower levels of CO and un-burnt hydrocarbon (HC) emissions. However, the higher levels of oxygen can lead to higher combustion temperatures and therefore to an increase in NO_x emissions. Specific emissions such as aldehydes can be expected to increase when burning ethanol compared to petrol or diesel (Brown, G. 2008).

In order to remain consistent with the approach used for the other vehicle fuels analysed in the present study as well as more generally in ecoinvent, we compiled emission factors for pure ethanol (E100) although, most commonly, ethanol is blended with gasoline at levels of to 85%. The factors arrived at are extrapolations of the work of (Martini, G. *et al.* 2009). They reported the effect from various blends of ethanol and petrol on a wide range of regulated and non-regulated emissions.

In **Figure 17** it can be seen that, due to the low energy content of ethanol the fuel consumption is significantly higher than using petrol but that because of the lower carbon content the CO_2 emissions are marginally lower. In addition to the absence of certain VOCs, PAHs and sulphur, it was assumed here that ethanol does not contain heavy metals. However, emissions of specific VOCs such as formaldehyde and acetaldehyde increase significantly with the switch from petrol to ethanol (Martini, G. *et al.* 2009).
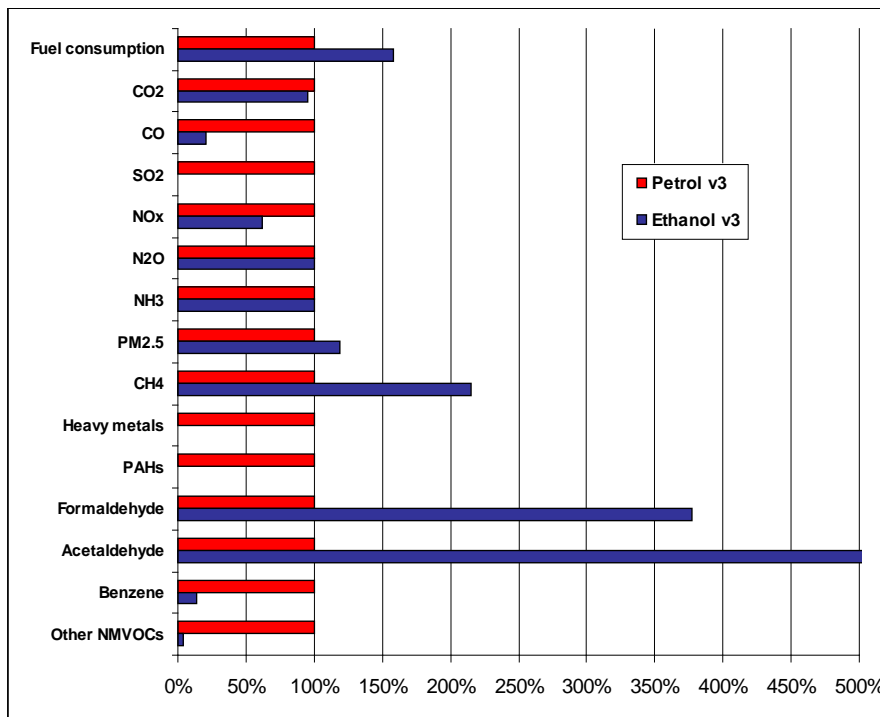
Figure 17: Comparison of fuel consumption and emissions for a mid-size passenger car of Euro3 standard fuelled with petrol (ecoinvent v3 dataset) and ethanol.

3.3.4. Synthetic crude oil (SCO) production from Canadian oil sands

Oil sands are deposits of bitumen coated sand which exist in quantities of great relevance to global oil demand and within tens of meters of the surface. A variety of processes have been developed for its extraction. Upgrading of the bitumen is then done in complex facilities which are often a part of a general petroleum refinery. For the present project we therefore selected a specific SCO production route for which the product system could be clearly defined and for which the need for allocation between co-products could be kept to a minimum. It is also representative of a modern process and thereby relatively energy efficient and with reference to the emissions caused. However, this specific process chain analysed cannot be regarded as representative for fuel production from oil sands in general worldwide, or for a kind of "average" for use of this feedstock.

Research was based on the Athabasca Oil Sands Project (AOSP) in the state of Alberta, with data being relevant for the year 2008. The AOSP includes the Muskeg River surface mine and extraction plant, the Corridor Pipeline (500km) and the Scotford Upgrader close to Edmonton. The analysis required extensive calculations and conversions of units such that it is not practical to describe here in fine detail. Data and process information was from various sources - the more significant of which we acknowledge in the summarized process and methodology description below.
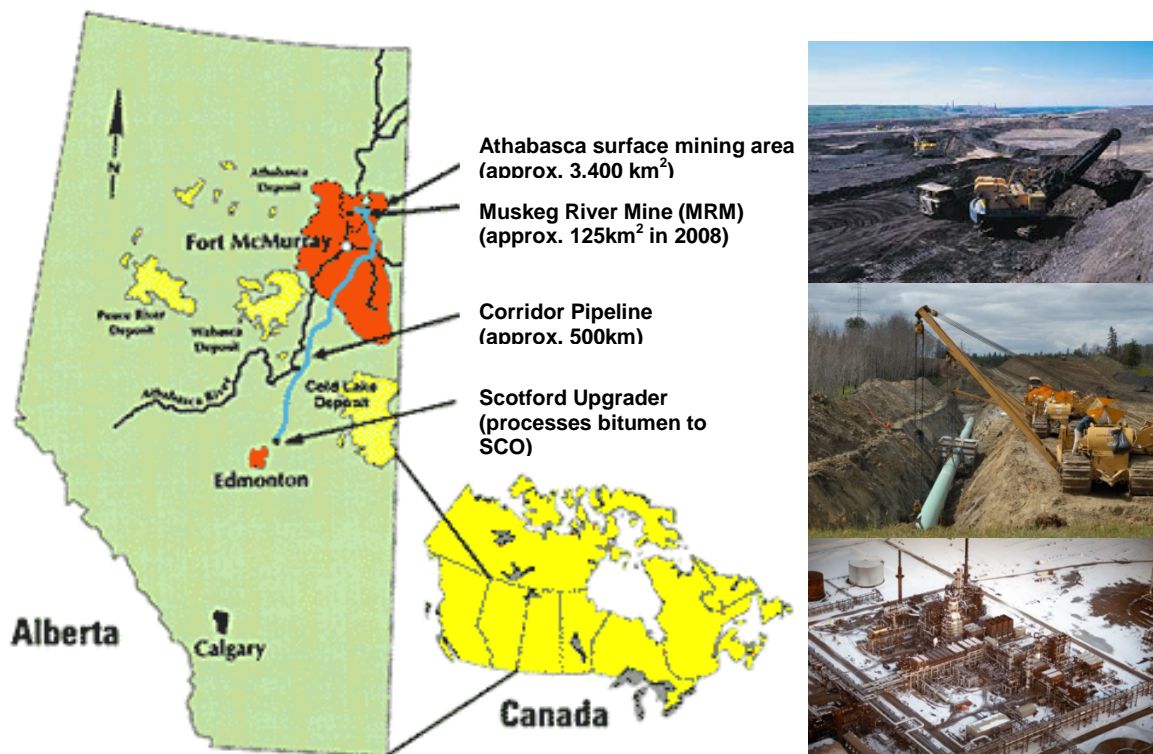
Figure 18:   Oil sands in Alberta, Canada, showing the main areas with deposits and the three main components of the Athabasca Oil Sands Project (AOSP).

Sources:

Map: http://bioage.typepad.com/photos/uncategorized/oilsands_map.gif

Mining: http://omiusajpic.org/files/2009/02/oil_sands_open_pit_mining.jpg

Pipeline: http://www.rbsomerville.com/admin/upload/116_projectPhoto1_lightbox.jpg

Upgrader: http://www.econbrowser.com/archives/2006/01/shell_upgrader.jpg

The process of oil sands mining, bitumen extraction, transport and upgrading is described in Figure 19.
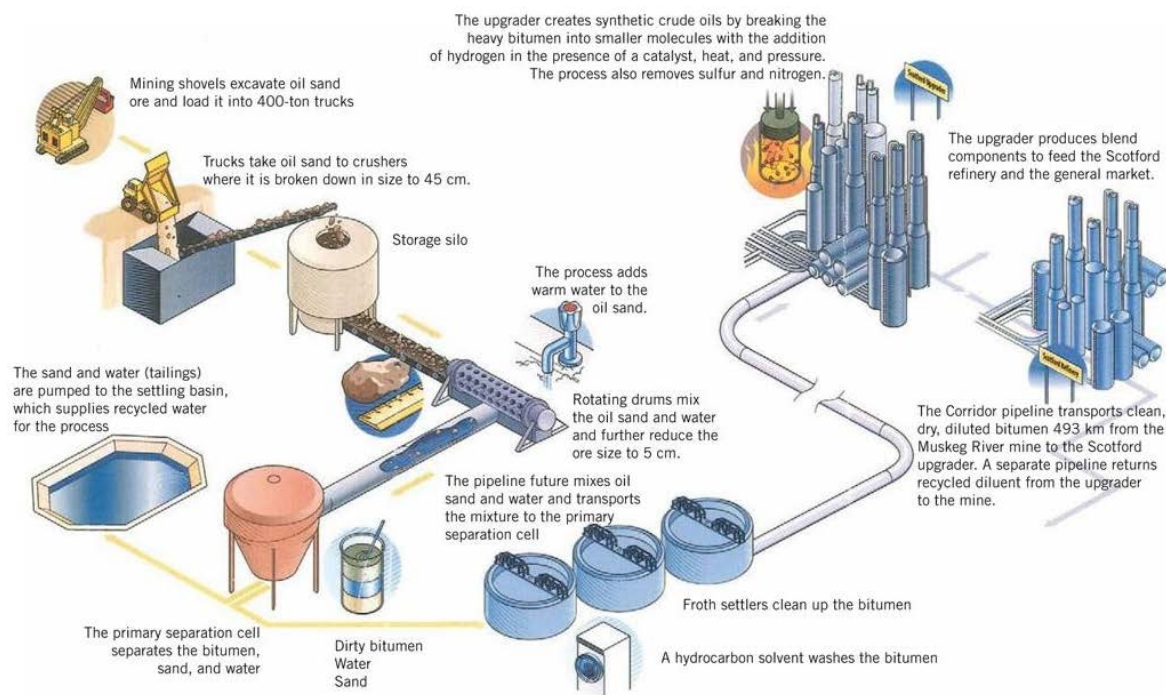
Figure 19 Process of producing SCO from oil sands in the AOSP.

Source: http://leadenergy.org/wp-content/uploads/2011/02/Athabasca-Diagram2.jpg

Mining and extraction

The mining and extraction of bitumen from oil sands is very clearly a practice which imposes significant burdens on the environment be it in terms of land use, resource use or emissions. Land use at the MRM has been calculated from satellite imagery by (Dean, A. *et al.* 2007) to have been around 4500 hectares (45km^2) in 2006 and that this was a growth of approximately 460 hectares on the previous year. By extrapolating this growth to 2008 and by using bitumen production quantities (Oil-Sands-Review 2009) the land use was calculated to be 0.17m^2 per barrel of bitumen (bbl bit). Surface mining is conducted where the oil sands are located within 75m of the surface (Shell-Canada 2009). The ore body itself contains around 11wt% bitumen, 85wt% sands & fines and 4wt% water (Allen, E.W. 2008) and modern extraction plants have a bitumen recovery efficiency of 95% (Keesom, W. *et al.* 2009). This results in approximately 1m^3 of oil sands per bbl bit. The ore body is excavated and transported in quarry trucks to the extraction plant as shown in Figure 19. Diesel use for this process is averaged from (Ordorica-Garcia, G. *et al.* 2007) and the GHGenius model (NRC 2008).

From Figure 19 it can be seen that bitumen extraction is achieved through washing and settling. The water for this process is largely recycled but also requires an extraction from the Athabasca River - calculated to be 0.6m^3 per bbl bit (Allen, E.W. 2008). The tailings, consisting of an aqueous solution of sand and fines, are stored on site in the form of tailings ponds which must be sealed to avoid contamination of the ground and water systems. Storage in the ponds is over a period of several years and allows for sedimentation and evaporation, and therefore for the separation of solids and the reuse of the water (Allen, E.W. 2008). The specific treatment of the solids in order to remove any levels of toxicity was not analysed in the present study and it was therefore not determined to what degree the eventual refilling of the surface mines with the solids contributes to potential impacts on the soil and water systems.

For all static demands for heat and power the MRM uses a 170MW natural gas powered CHP. The heat demand is larger than the electricity demand meaning that a proportion of the latter is exported to the grid and that it is assumed that the size of the CHP plant is designed to suit the heat demand. Only rough proportions of 60% on-site and 40% exported could be determined, resulting in 33kWh$_{el}$ per bbl bit. Sodium hydroxide is used to help separate the

bitumen for which an informed estimate of 3.2kg per bbl bit was used. Emissions from all major industrial facilities must be reported to the Canadian authorities and which are then made public. These are distinguished as greenhouse gas (GHG) emissions and other pollutant emissions to air, water and soil available from (Environment-Canada 2009a). Emissions from the CHP plant were subtracted from the reported values to avoid double-counting. Infrastructure demands were calculated using the current production volumes and a lifetime of 30years. The infrastructure datasets themselves were adapted from those for the surface mining of lignite in Germany found in ecoinvent v2.1.

Bitumen transport

Once extracted from the other ore substances the bitumen is diluted with naphtha in order to facilitate transport by pipeline. This mixture is known as dilbit (diluted bitumen). The naphtha is recovered at the upgrader and returned to the extraction plant. The electricity demands for transport were calculated on the basis of the specific gravity (API°) of the naphtha, bitumen and dilbit resulting in 0.01kWh/tkm naptha and 0.027kWh/tkm dilbit. Electricity is from the medium voltage Canadian supply mix.

Bitumen upgrading

The inputs, flows and outputs at the upgrader were calculated based largely on the work of (Keesom, W. *et al.* 2009) as well as the emissions data from (Environment-Canada 2009b). For the purposes of this study and its contextual focus on the use of transport fuels an Europe, the upgrader was assumed to only produce one quality of SCO for further refining to petrol and diesel. The Scotford Upgrader represents a modern facility which makes use of technologies such as on-site CHP and ebullated-bed hydrocracking. The latter is a catalytic process operating at high temperature and pressure in order to break apart the long and heavy hydrocarbon molecules and to separate impurities. The lighter hydrocarbons are then recombined with purified hydrogen (produced on-site from natural gas) to produce a high quality SCO. More traditionally, bitumen has been upgraded using purely thermal and therefore energy intensive processes in order to break the molecules and to extract surplus carbon or "coke" as a waste by-product. The ebullated-bed method adds hydrogen rather than extracting the coke and therefore makes more efficient use of the bitumen supply: here calculated to be 1.02 bbl SCO per bbl bit. Apart from bitumen the other main inputs to the upgrading process are natural gas for both the production of hydrogen and for fuelling the CHP plant, catalyst material and water. Other fuel gases used are produced in the refinery as by-products. Natural gas for fuelling the CHP plant at the Scotford Upgrader is modeled in a similar way to that for the MRM except that 70% of electricity is used on-site as opposed to 60% at the MRM. Electricity demand at the upgrader is thus 32kWh per bbl SCO. According to (Galiasso-Tailleur, R. 2007) the catalyst material is composed of 12.5wt% molybdenum trioxide (MoO_3), 4.2wt% nickel oxide (NiO) and the rest of aluminium oxide (Al_2O_3). Catalyst demand of 0.75kg per bbl SCO was derived from (McKnight, C. *et al.* 2007). Water consumption was taken from (Shell-Canada 2009) and calculated to be $0.07m^3$ per bbl SCO. As mentioned above, emissions from the Scotford Upgrader were those reported to (Environment-Canada 2009b) with those from the CHP plant being deducted from these. The upgrader facility is assumed to have a lifetime of 30yrs and adopts the dataset for onshore crude oil production from the ecoinvent database v2.0.

SCO transport and refining

Transport of the SCO from Alberta to the eastern Seaboard of Canada by pipeline and from there to Europe via tanker was modelled by adapting the currently available dataset for crude oil produced in South America and transported to Europe. From Table 18 it can be seen that the composition of the SCO is very similar to that of crude oil currently existing in ecoinvent (Jungbluth, N. 2007). It was therefore decided that the further refining of SCO would not require significantly different treatment and that all further processing of SCO could be

considered the same. The production of low sulphur petrol and diesel from SCO therefore substitutes inputs of crude oil with that of SCO one-to-one.

Table 18: Composition of the synthetic crude oil (SCO) analysed in this study in comparison with that of average crude oil from the ecoinvent database v2.0 (Jungbluth, N. 2007).

	Unit	Ecoinvent	Jacobs (2009)	Wang Jun et al (2007)	This study
		Crude oil	SCO Eb-bed*	SCO Average	SCO Eb-bed
C-content	Wt %	83 - 87		87.5	**87.3**
H-content	Wt %	12		12.3	**12.2**
O-content	Wt %	0.05 - 1.5			
N-content	Wt %	0.1 - 2.0	0.12	0.08	**0.12**
S-content	Wt %	0.05 - 6.0	0.35	0.14	**0.35**
Density	kg/l	0.86	0.92	0.87	**0.92**
API (Specific gravity)	°API	22	22.1	29.8	**22.1**
LHV	MJ/kg	43.2	39.6		**40**

* Ebullating bed – in reference to the specific upgrading technology used.

Combustion of petrol and diesel from SCO in road transport vehicles

The composition of petrol and diesel from SCO is considered to be the same as from conventional sources (see **Table 15**) and therefore does not lead to differences in fuel use or emissions from vehicle operation.

3.4. Production and use of gaseous fuels

This section deals with the conversion of biomass substances to gases of specific methane content. As a rule-of-thumb, gases with less than 95vol% CH_4 are not of equivalent quality to natural gas and are therefore a form of biogas. Biogases tend to have a CH_4 concentration of 50 to 75%. Above 95vol% methane content the gas can be considered a bio-methane or, more generally, a synthetic natural gas (SNG). There are many sources of biogas or SNG and an increasing number of ways of producing them – not all of them suitable to application in Switzerland. In this study we consider agricultural crops and by-products, industry waste and human waste as sources of biogas or SNG. Alfalfa grass is bio-digested to biogas which can be burned in a specially adapted ICE CHP, and in a further scenario this biogas is purified to SNG – also for use in an ICE CHP. Due to the quantities and locations of the gas products they are not considered for any further applications. Biogas from sewage treatment facilities is purified on-site using the amine washing "biomass-CO_2-methane" (BCM) process (DGE 2004) and not combusted as biogas. The hydrothermal gasification process maintains the feed flow in aqueous form with methane being separated by physical adsorption. A biogas energy carrier is therefore not produced from the process.

3.4.1. Biogas and SNG from alfalfa grass

The research started with an analysis of the composition of alfalfa grass and compared this with the grass compositions used to form the grass mix currently in ecoinvent (Jungbluth, N. *et al.* 2007). As can be seen in Table 19, the composition of alfalfa grass is not sufficiently different from that of the grasses which contribute to the ecoinvent dataset for biogas from grass (which is mainly made from intensive meadowland cultivation) such that the further conversion of alfalfa to biogas will be a more or a less intensive process. The new dataset for this process therefore simply involved the replacement of the biomass source.

Table 19: Properties and compositions of grass crops for the production of biogas. Data taken from ecoinvent (Jungbluth, N. *et al.* 2007) and the BIOBIB database (Reisinger, K. *et al.* 1996).

| | Unit | Ecoinvent | | This study |
		Grass, extensive meadows	Grass, intensive meadows	Alfalfa grass
Dry matter content	%	15	15	15*
LHV	MJ/kg			17.3
HHV	MJ/kg	18.9	17.9	18.4
Carbon	wt% dry matter	45.0	45.0	45.9
Hydrogen	wt% dry matter	5.0	5.0	5.2
Nitrogen	wt% dry matter	2.4	2.4	2.5
Chloride	wt% dry matter	0.8	0.8	0.4
Sulphur	wt% dry matter	0.013	0.013	0.18
Ash	wt% dry matter	9.35	9.35	9.14
Oxygen	wt% dry matter	37.5	37.0	36.7

* Value not given in the source, therefore assumed to be this based on the similarity of the heating values and other fractions of substances.

The composition of the biogas from alfalfa is shown in Table 20.

Table 20: Compositions and properties of biogases in ecoinvent of relevance to the present study (Jungbluth, N. *et al.* 2007) and of biogas from alfalfa grass as determined in the present study. Values in italics have been calculated based on the biogas mix contributions of 8.6% from biowaste and 91.4% from sewage.

| | | Ecoinvent | | | This study |
	Unit	From waste	From sewage	Biogas mix	From alfalfa
Methane CH_4	Vol %	67	63	63.3	**55**
Carbon dioxide CO_2	Vol %	32.1	33.6	33.5	**45**
Nitrogen N_2	Vol %	0.7	3.4	*3.2*	
Oxygen O_2	Vol %	0.25		*0.02*	
Hydrogen sulphide H_2S	Vol %	0.0005			**<0.01**
Density	kg/Nm^3	1.12	1.15	1.15	**1.28**
HHV	MJ/Nm^3	26.6	25.0	*25.1*	**21.9**
LHV	MJ/Nm^3	24.0	22.6	22.7	**19.7**

Two scenarios for the use of the biogas from alfalfa grass were analysed: direct combustion in a small CHP unit and purification to SNG. Direct combustion used the same system currently existing in ecoinvent and fuelled by the biogas mix, representing a 160kW internal combustion cogeneration unit (Heck, T. 2007). The changes made were to increase the biogas demand according to the LHV and the CO_2 emissions according to the carbon content of the biogas. NO_x emissions were kept the same because at this temperature of combustion these depend far more on nitrogen from the air rather than nitrogen in the biogas. All other emissions were kept the same due to the uncertainty in the parent dataset.

Purification of the biogas to SNG follows the conventional route of pressure swing adsorption (PSA) also used in ecoinvent for the purification of the biogas mix. In this process the CO_2 is adorbed at high pressure onto the surface of activated carbon and then extracted through reduction of the pressure. The process also removes sulphur compounds in order to purify the quantity of CH_4 in the product gas or SNG. For PSA of the alfalfa biogas the inventory flows were calculated based on the compositional differences to that of the existing biogas

mix. The inventory data for biogas upgrading are given in Table 21 and the properties and composition of the SNG in Table 22.

Table 21: **Datasets for the production of SNG (1 Nm³) according to ecoinvent (Jungbluth, N.** **_et al._ 2007) and the specific to the current study.**

		Unit	Ecoinvent	This study
			SNG, from biogas mix	SNG, from alfalfa biogas
Materials / fuels	Biogas, production mix	m³	1.5	
	Biogas, from alfalfa	m³		1.727
	Electricity	kWh	0.5	0.576
	Chemical facilities	kg	4E-11	4E-11
Emissions to air	Carbon dioxide, biogenic	kg	0.87	1.16
	Methane, biogenic	kg	2.23E-02	2.56E-02
	Hydrogen sulfide	kg	3.49E-06	4.02E-06
	Sulfur dioxide	kg	5.52E-04	6.35E-04
	Heat, waste	MJ	1.8	2.07

3.4.2. SNG from hydrothermal gasification of animal slurry and wet wood

By operating at high temperature (>374°C) and high pressure (>221 bar), the process exploits the supercritical point at which the water in the biomass decomposes the larger organic molecules and also leads to the precipitation of inorganic salts which can then be separated and made available as a by-product e.g. as a fertiliser. A catalyst is then used to form methane within the aqueous feed. As methane is virtually insoluble in water, separation of the methane can then occur using physical adsorption, leading to the products of SNG, water and CO_2 (Luterbacher, J. _et al._ 2009; Gassner, M. _et al._ 2011).

The two feedstocks considered for use in plants located in Switzerland were animal slurry (as waste product e.g. from pig farms) and residues from forestry and sawmills e.g. wet sawdust. Life cycle data for the production of the SNGs was taken from the study by (Luterbacher, J. _et al._ 2009) such that the analysis for the present study focused on the end-use combustion. The properties and composition of the SNGs are given in Table 22.

3.4.3. SNG from sewage biogas using the BCM process

As shown in Table 20, the biogas extracted at the sewage plant consists of almost two-thirds CH_4. An alternative method for the upgrading of biogas to SNG has been developed by (DGE 2004) in which an aqueous amine solution absorbs the CO_2 from the cleaned and dried biogas stream. The process occurs at atmospheric pressure and low temperature (40°C) and results in a CH_4 purity of 99% (MT-Biomethan 2011). The CO_2 containing wash solution is then heated and put through a separator in order to regenerate the pure amine solution and to release the CO_2. A description of the process is given in Figure 20.

The inventory data for the BCM process were taken from (EMPA 2009) and the properties and composition of the SNG are given in Table 22.
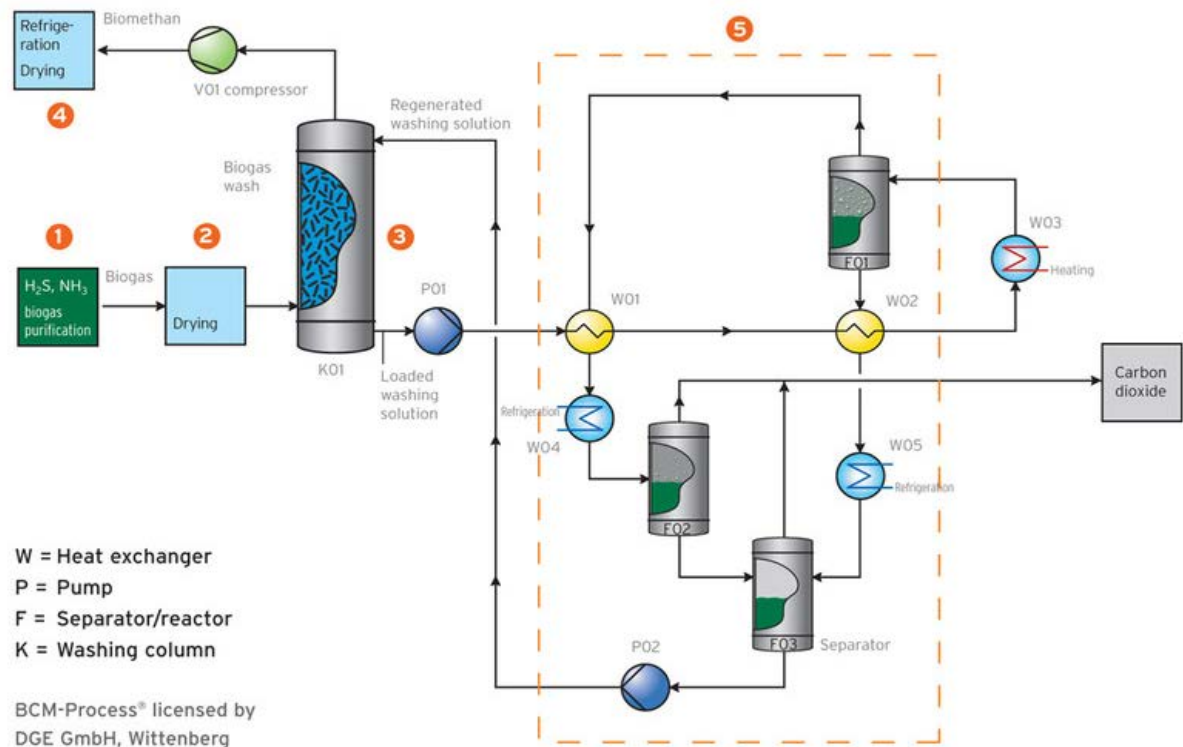
De-sulphurisation
Drying
Amine wash (atmospheric pressure)
SNG cooling and drying
Amine solution regeneration

Figure 20:   Process flow diagram of the BCM process for purifying biogas (MT-Biomethan 2011).

For a gas to be considered of equivalent quality to natural gas it should meet specific requirements in terms of its energy density, sulphur content as well as fractions of other impurities (SVGW 2004). Table 22 gives the properties and compositions of the SNGs considered in the present study, as well as those of natural gas from the ecoinvent database (Faist Emmenegger, M. *et al.* 2007) for comparison.

Table 22: Compositions and properties of the product gases in comparison with those of natural gas from the ecoinvent database (Faist Emmenegger, M. *et al.* 2007).

		Ecoinvent	This study			
		Natural gas	Synthetic natural gas (SNG)			
			Hydrothermal gasification		BCM	PSA
	Unit	EU-mix	Cattle slurry	Wet wood	Sewage biogas	Alfalfa bio-gas
CH_4	Vol %	94.4	95	97	99	96
CO_2	Vol %	0.6			1	2
H_2	Vol %		5	3		
N_2	Vol %	1.7				1
O_2	Vol %					1
H_2S	Vol %	<0.01	<0.01	<0.01	<0.01	<0.01
Alkanes	Vol %	3.3				
Density	kg/Nm^3	0.76	0.69	0.7	0.73	0.75
HHV	MJ/Nm^3	40.2	38.6	39.1	39.6	38.2
LHV	MJ/Nm^3	36.5	34.6	35.1	35.5	34.5
LHV	MJ/kg	48.0	50.1	50.1	48.6	46.0
Wobbe index	MJ/Nm^3	47.6	47.6	47.8	47.3	45.2 (-4%)

3.4.4. SNG combustion

As shown in Table 11 and as described in the summary, the analysis considered a variety of end uses for the SNGs. For all except that produced from alfalfa grass, the combustion of SNG was analysed in small and large-scale stationary facilities as well as in passenger vehicles. It was judged that SNG from alfalfa grass would be only potentially available in relatively small quantities for realistic consideration of combustion in large power plants or in passenger cars.

Due to the required quality of SNG its combustion characteristics will only vary from those of natural gas as a result of the slight variations in its composition. In this study we determine SNG demand according to the individual fuel's lower heating value (LHV) in relation to that of natural gas. Whilst CO_2 emissions will depend on the carbon content of the SNG, other GHG emissions of CH_4 and N_2O will remain the same and due to similar combustion temperatures the NO_x emissions will also remain the same.

3.5. Update and parameterisation of incineration plants

The life cycle inventories for the incineration of biomass in an average Swiss municipal solid waste incinerator (MSWI) were updated and expanded. Current data for net energy gain, DeNOX technology and dioxin emissions were applied. The inventory model was extended to calculate energy consumption according to causing factors and attribute incomplete combustion products according to waste fraction properties instead of a constant per kg waste amount.

3.5.1. Gross energy efficiency

Waste incinerators can produce usable heat and electricity from the incinerated waste. Boilers produce steam, which can be fed into district heating systems. Steam can also be converted into electricity with steam turbine generators and fed into the electricity grid.

On average operation, Swiss MSWI convert 15.37% of the lower heating value of the input waste into electricity and 26.91% into usable heat (data for 2009 from (BFE 2010)). The remainder of 57.7% is waste heat.

Figure 21 shows the development of gross energy efficiencies of individual Swiss MSWIs for the years 2000, 2002 and 2006 (data from (BAFU 2011)). On average (cf. Swiss flag symbol in Figure 21), there has been a shift towards more electricity production, while heat

production remained approximately constant. Heat production and distribution is commonly restricted by the existence of suitable heat consumers within a reasonable distance, while an increased electricity production can easily be fed into the grid. In 2006 all plants produce at least some electricity, while earlier there were two "heat-only" plants, Lausanne and Zermatt. In 2006 there are five plants of 29 which produce only electricity and no or very little usable heat.
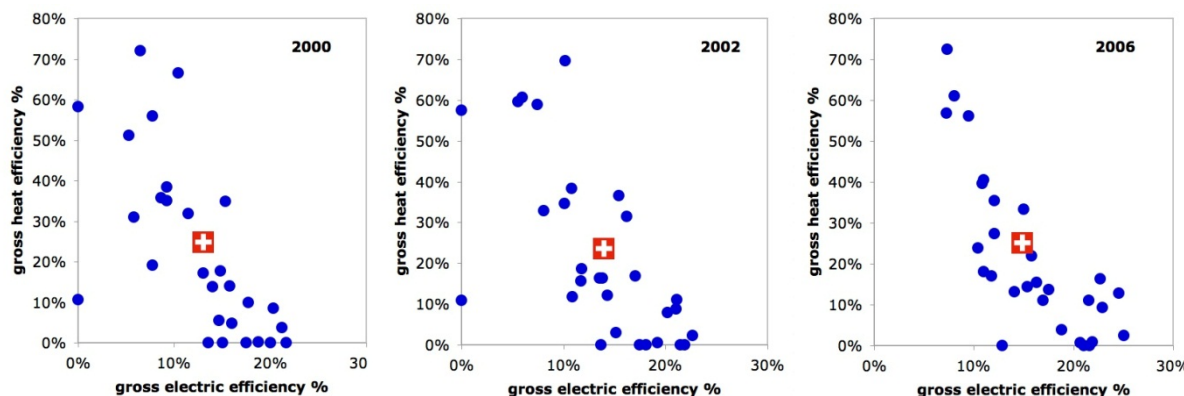
Figure 21: **Development of gross energy efficiencies of individual Swiss MSWIs for the years 2000, 2002 and 2006**

3.5.2. Internal energy consumption

Some of the produced energy is needed to run the incinerator operation. This internal energy consumption of heat and electricity is deducted from the gross energy production of a waste and results in the net energy production. For average operation, Swiss MSWIs have positive net energy production, i.e. no external energy supply is needed[2].

In former ecoinvent datasets of municipal incineration, the internal energy consumption was simply a constant amount per kilogram of waste. This is changed here to a variable amount depending on the waste characteristics and causalities for internal energy consumption.

For instance, a large part of internal electricity consumption is caused by suction fans of the flue gas purification stage. Accordingly, a waste which does produce no or little raw flue gas needs less internal electricity than an average waste. Similarly, internal heat is needed to heat up combustion air. An inert, unburnable waste will need no combustion air and therefore no internal heat, as opposed to a burnable waste. Table 24 shows the distribution of internal energy demand to causes.

---

[2]   Apart from natural gas needed to heat up the flue gas in the DeNOx stage.

Table 23: Contribution to internal energy demand in MSWI

Internal electricity demand [1]	%	Waste-related cause
Flue gas treatment	46.4%	Raw flue gas volume [2]
Steam generator	20.6%	Lower heating value
Combustion chamber	11.3%	Combustion air input [3]
Waste handling and shredding	7.2%	Waste mass input
Auxiliary facilities	7.2%	Waste mass input
Building services	5.2%	Waste mass input
Wastewater treatment	2.1%	Mass arriving in wastewater treatment
Internal heat demand [4]		**Waste-related cause**
Air pre-heating	100%	Combustion air input

1   Data from (Egli, S. 2005). 3% for sludge incineration was distributed pro rata.

2   Largest originators are flue gas suction fans, electro-static precipitation filters (EPS) and pumps for wet flue gas treatment (Egli, S. 2005), p.15/16).

3   Largest originator is the combustion air processing ((Egli, S. 2005), p.14).

4   Any internal heat demand is attributed here to combustion air pre-heating

Table 24: Distribution of internal energy demand to causes

		Internal energy demand per kg average waste	Allocands in average waste		internal energy factors
Electricity		kWh/kg	unit /kg waste	unit	kWh/unit
Internal electricity demand [1]	**100%**	**0.1274**			
Flue gas treatment	46.4%	0.05909	7.48	m^3 raw gas volume	0.007896
Steam generator	20.6%	0.02623	11.74	MJ LHV	0.002234
Combustion chamber	11.3%	0.01439	6.75	m^3 combust air input	0.002133
Waste handling and shredding	7.2%	0.00917	1	kg waste input	0.009169
Auxiliary facilities	7.2%	0.00917	1	kg waste input	0.009169
Building services	5.2%	0.00662	1	kg waste input	0.006622
Wastewater treatment	2.1%	0.00267	0.0057	kg is wastewater treatment	0.468817
iron descrapping	–	–	–	kg metallic iron in slag [2]	0.04167
Heat		MJ/kg			MJ/unit
Internal heat demand [3]		**0.4039**			
Air pre-heating	100%	0.4039	6.75	m^3 combust air input	0.05986

1   (BFE 2010: Tab. 27), annual incinerated waste mass from (BAFU 2010)

2   0.04167 kWh per *kg metallic iron in slag*, corresponds approximately to 0.00083 kWh/kg average waste.

3   (BFE 2010: Tab. 27), but to compensate for non-reporting MSWI applied a factor of 1.5 based on BAFU statistics for 2000, 2004, and 2006[3]. Annual incinerated waste mass from (BAFU 2010).

3.5.3. Biowaste composition

Water content

In a literature study the water content of biowaste was researched for this project. The considered sources were (Kaiser, E.R. 1966; Kaiser, E.R. 1975; Bilitewski, B. *et al.* 1991; Tillman, D.A. 1991; Mark, F.E. *et al.* 1994; Maystre, L.-Y. *et al.* 1994; BUWAL 1995; Zimmermann, P. *et al.* 1996; Schleiss, K. *et al.* 2000; Edelmann, W. *et al.* 2001; Meraz, L. *et al.* 2002; Rolland, C., Scheibengraf, M. 2003; Williams, R.B. *et al.* 2003; Banks, C. 2004; Angele, H.-C. 2006; Biomasse Schweiz 2006; Entsorgung + Recycling Zürich 2006; Wellinger, A. *et al.* 2006; Jayalakshmi, S. *et al.* 2007; Jungbluth, N. *et al.* 2007; Kranert, M. *et al.* 2007; Edelmann, W. 2008; Thitame, S.N. *et al.* 2010).

Generally the water content of biowaste varies considerably, even for single fractions. Edelmann (2008) finds an average value of 69.3%, with a 58.3% – 80.3% confidence interval. Food wastes from kitchen, restaurants or food processing often have high water contents. Woody garden waste has often lower water content; fresh grass waste has frequently higher water content.

Edelmann (2008) investigated the waste for digestion and composting plants. Not all this waste is also suitable for incineration. Liquid waste from commerce (e.g. salsa and soup) can increase water content, while garden waste can also include earth and soil from potted plants, with rather low water content. Edelmann finds that especially the presence of industrial food waste from restaurants or food processing can feature relatively high water contents of above 75% (p.11). For kitchen and garden waste with little or no industrial waste, Edelmann finds a water content of 66-67% to be typical.

For the present study 3 different biowaste compositions are investigated:

1)   Average biowaste with 65% water content (mean of all literature values)

2)   Average biowaste with 60% water content (mean for garden/park/yard waste)

3)   Average biowaste with 72% water content (mean for food/kitchen waste)

Only the water content is varied. The composition of dry matter is assumed to be identical.

---

3   A significant part of MSWI plants in the plant-specific BAFU statistics report *no* internal heat demand, which is considered unrealistic, i.e. should be read as "not available" (statistics see (BAFU 2011)). The sum of the demand reported in BAFU statistics equals the figure given in BFE statistics since 2003 (before that date, BAFU had significantly higher figures). When heeding only plants reporting above zero, a specific internal heat consumption per kilogram waste results, which is approximately a factor 1.5 larger than a value considering also plants with a zero value. This factor is applied to the internal heat demand figure reported in (BFE 2010), to compensate for the zero-reporting plants.
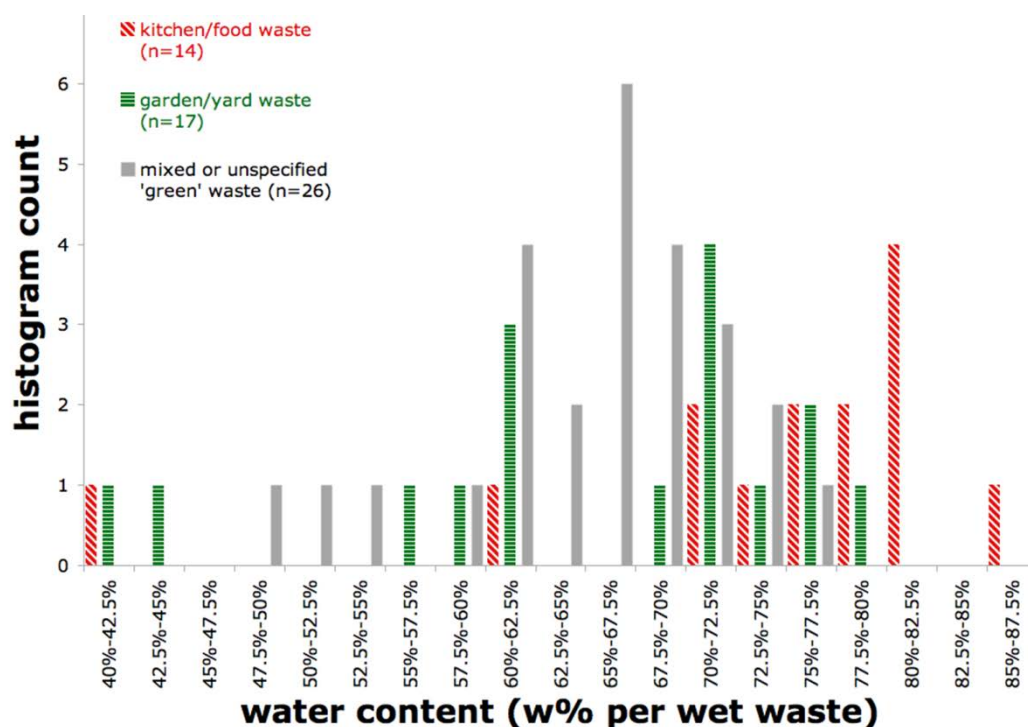
Figure 22: Histogram for water content in biowaste. Results of the literature study.

3.5.4. Dioxin Emissions

In the past, especially in the 1970s and 80s, Swiss MSWI have been the dominant source of dioxin emissions in Switzerland (BUWAL 1997). Swiss MSWIs have much improved in this respect. The low dioxin emissions of Swiss MSWI have been further reduced in the last decade. Dioxin emissions factors are mostly determined by combustion conditions. Elevated temperatures and homogenous combustion conditions reduce the probability of dioxin formation. Data from German MSWIs suggests however that there is considerable variation in dioxin emissions factors over three orders of magnitude reflecting a wide range of combustion conditions (Löschau, M. 2009: Tab. 3).

In 2007 the Swiss Federal Air Pollution Control Regulation (LRV) introduced a new threshold limit value for dioxins for waste incineration of 0.1 ng I-TEQ/m^3. Current major source of dioxin emissions are now households (illegal burning of waste, wood furnaces and fireplaces).

The historical pollution profile has led to a persisting bad image of the incinerators, especially in the environmental community of the USA. While it is probable that inferior MSWI performances regarding dioxins still exist in the world, for the current study which focuses on Swiss MSWI, the observed low emission factors for Swiss plants are employed.

Literature values of dioxin measurements in Swiss MSWIs are compiled in Table 25:. For average operation with average municipal waste a value of 0.01 ng/m^3 is chosen for this study[4]. This value also matches the one given by Hans-Peter Fahrni, director at BAFU for waste and resources, for modern MSWIs (Fahrni, H.P. 2006). With an average flue gas volume (dry, 11%O2) of 6.44 m^3 per kg average municipal waste an emission factor of 0.0644 ng per kg average waste results. This value is further processed depending on waste fraction characteristics as outlined in the chapter 'Incomplete combustion products' below.

---

[4]   For plants without any DeNOx process a value of 6 ng/m^3 is retained. Although that option is not part of the current *Swiss* MSWI technology mix, it might be relevant in other regions.

Table 25:   Recent Dioxin Measurements in Swiss MSWIs

MSWI location		ng/m^3	Date
GL Niederurnen Line 1	1	0.005	Aug 2007
GL Niederurnen Line 2	1	0.018	Nov 2008
LU Luzern	2	0.00025	2009
SO Zuchwil	3	0.03	2009
TI Giubiasco Line 1	4	0.00925	Jan 2010, test run
TI Giubiasco Line 2	4	0.00467	Jan 2010, test run
AG Turgi	5	0.0225	5th May 2010
VD Tridel Line 1	6	0.008	15th June 2009
VD Tridel Line 2	6	0.013	15th June 2009
This study		**0.01**	

1   http://www.kva-linthgebiet.ch/Messwerte.htm (18 Feb 2011)
2   http://www.kva-luzern.ch/index.php?id=20 (18 Feb 2011)
3   http://www.kebag.ch/cms/front_content.php?idcat=46 (18 Feb 2011)
4   http://www3.ti.ch//DT/cartellastampa/pdf-cartella-stampa-43376492648.pdf (18 Feb 2011)
5   http://www.kvaturgi.ch/fileadmin/pdf/Emissionen_2010.pdf (18 Feb 2011)
6   http://www.tridel.ch/userfiles/pdf/TRIDEL_2008-09-10_Comparatif_emissions.pdf (18 Feb 2011)

3.5.5. Process-specific emissions

In ecoinvent 2000, various air emissions, like NMVOCs, benzene, thermal NOx or dioxins, were attributed to be *process-specific*, i.e. per kilogram of waste a *constant* amount of these emissions were inventoried. This modelling was refined for this study.
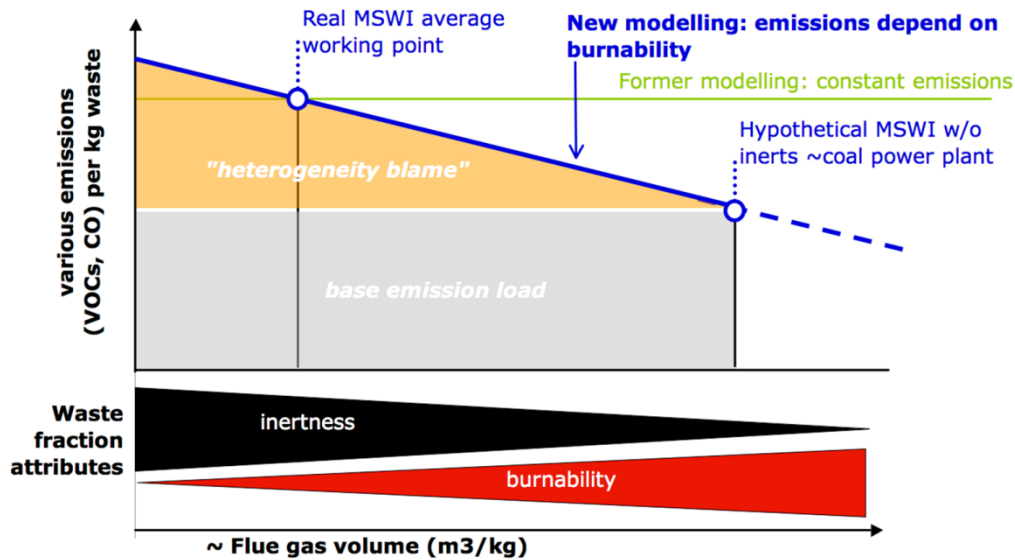
Incomplete combustion products

Emissions of carbon monoxide CO and organic compounds (NMVOCs, benzene, dioxins and others) originate from burnable materials, which are incompletely combusted due to heterogeneities in all or either of the following: combustion temperatures, fuel/oxygen mixing ratios, fuel residence times (see e.g. Nussbaumer, T. 2006: 180ff). These heterogeneities can result from combustion technology and process conditions alone, but inert incombustible waste material – like glass, metals, bones or ceramics – amplifies their occurrence. So the emission of incomplete combustion products in average incinerator operation are to be blamed on the occurrence of combustible waste on one hand, but to a certain extent simultaneously also on the presence of inert materials and also applied technology. This study seeks to inventory emission attributable to the combustion of distinct waste fractions, like one kilogram of plastic or one kilogram of tin sheet. So how can the emissions of incomplete combustion products from average MSWI operation be attributed to a specific fraction?

For this study, a generic estimate was attempted, how low the emissions could be, if only well burnable waste were incinerated. A comparison with emissions from hard coal power plants was used, based on mass of fuel input[5] ((Dones, R. 2007), Tab 9.48, p.138). For relevant VOC species and CO it was found that the coal power plant has only 44% to 85% of the emission levels of a waste incinerator. A value of 62% for sum NMVOC is found, which is practically identical to the unweighted arithmetic mean for all species. This finding is taken to imply that if a waste incinerator would only incinerate burnable waste without any inert materials like glass or bulk metals, the emission levels of organic compounds and CO would be 62% compared to average waste. With higher burnability of the waste lower VOC emissions are attributed; while higher inertness – low content of burnable material – bears higher VOC emissions. The characteristics of inertness vs. burnability of a waste are taken to

---

5   With the metric "kg emission per kg fuel" the air excess number has no influence.

be represented by the amount of *flue gas volume* a particular waste fraction causes from combustion (with the unit m³ gas/kg waste). The scheme below summarises the new modelling of incomplete combustion products.

Figure 23:   New modelling approach for incomplete combustion products.

The new model is based on two points marked with blue circles: The emission level from average MSWI operation (left, 100% emission level)[6] and the reduced emission level when only burnable material is incinerated (right, 62% emission level). The waste characteristic of "burnability" is translated into the metric "specific flue gas volume" (x-axis at the bottom): for the left point the flue gas volume from average MSW is taken (7.48m³/kg waste); for the right point flue gas volume from combustion of hard coal in MSWI is taken (14.15m³/kg coal)[7]. A simple linear relationship through these two points assumed, representing the decrease of attributable emissions with increasing burnability[8]. The inventoried emissions can be thought of being caused by a base emission (grey box *"base emission load"*) and a variable component representing the inertness or burnability of a waste fraction (orange triangle *"heterogeneity blame"*). The level of emissions attributable to an investigated water fraction within average MSWI operation is calculated with the following formula:

$$emission\ level = 62\% + \frac{V_c - V_i}{V_c - V_m} \cdot (1 - 62\%)$$

where

V_c = specific flue gas volume for hard coal, 14.15 m³/kg

V_i = specific flue gas volume for the investigated waste fraction, in m³/kg

V_m = specific flue gas volume for average municipal solid waste, 7.48 m³/kg

---

[6]   This level in turn is influenced by the technology mix of DeNOx stages (SNCR, SCR, or none), taken to represent the generic MSWI plant. This heeds the fact that generation and emission of incomplete combustion also depends on MSWI technology, not only waste characteristics (see chapter 'Volatile organic compounds VOC' in ecoinvent report 13-II (Doka 2003)).

[7]   These figures include an air excess number (Λ lambda) of 2, appropriate for MSWIs.

[8]   At the other end of the spectrum, a completely inert waste fraction, like glass, receives an elevated level of emissions of about 143% of the average MSWI operation level. Keep in mind that the inventory represents the emissions attributable to a specific waste fraction, when burned in an average MSWI *together with average municipal solid waste*. So a specific flue gas volume of zero does not mean that *all* the input to the incinerator is inert. Burnable materials are co-incinerated with the investigated inert waste fraction. The latter causes more heterogeneities and is attributed an elevated level of emissions.

Hard coal has a fair amount of ash (8 w%), so it still has some residual "inertness". Within the MSWI model, waste fractions with even larger flue gas volumes are possible. Polyethylene has the largest specific flue gas volume of the investigated fractions (21 m³/kg, 2.4 w-% ash). It therefore receives only 20% of the average MSWI emission levels of VOC and CO species.

$$62\% + \frac{14.15 - 21.5}{14.15 - 7.48} \cdot \left(1 - 62\%\right) \quad = 62\% + (-1.1) \cdot 38\% \quad = 62\% - 42\% \quad = 20\%$$

Thermal NOx

Thermal NOx are nitrogen oxides in combustion flue gas which originate from the reaction of elemental nitrogen N2 in air during the combustion process (as opposed to *fuel-NOx* which come from combusted nitrogen in the fuel). Thermal-NOx generation depends entirely on combustion air and elevated temperatures. In this study, the contribution of thermal NOx and the expenditures for removal in DeNOx processes[9] for the treatment of a particular waste fraction is assumed to be proportional to the input of combustion air necessitated by that waste fraction. For average MSW a value of 6.748 m³ combustion air input per kg is calculated, including an air excess number Λ lambda of 2.

Waste fractions with larger combustion air input will have proportionally larger burdens from thermal NOx, waste fractions not requiring combustion air, like glass, will have no burdens from thermal NOx. In the inventory, the emissions and expenditures from thermal NOx are summed up with those for *fuel-NOx*, and are not devised separately.

3.5.6. DeNOx technology mix

The MSWI model distinguishes four different kinds of DeNOx-Technologies. For the present inventory a Swiss average technology mix is assumed. The technology mix was updated to figures derived from 2007 data given in (BAFU 2008: 102ff.). A mix weighted with waste treatment capacity was calculated. The figure for SCR (75.5%) was divided according to the former split into low-dust SCR and high-dust SCR.

Table 26: Former and New DeNOX technology mix in the MSWI model.

	Without Denox	SNCR	SCR-high dust	SCR-low dust
ecoinvent v1.0-2.2 representing year 2000	13.8%	29.4%	32.2%	24.6%
This study representing year 2007	0.0%	24.5%	42.8%	32.7%

The different DeNOX technologies influence the final air emissions of NOx, ammonia NH_3, nitrous oxide N_2O, as well as the auxiliary inputs of natural gas, ammonia, and catalyst materials ($TiO_2 + V_2O_5$). Waste-specific emissions depend on the amount of nitrogen entering the flue gas, while emissions connected to thermal NOx depend on the waste-specific combustion air input.

3.5.7. Allocation

The economical revenue stream for MSWI is chiefly the disposal fee. Additional revenue can be generated by sold net energy. As the delivered products "disposal function" and "energy generation" share no sensible physical parameter, an allocation with an economical key, i.e. the market value of the generated products, is performed here[10].

---

[9]   Expenditures from DeNOx are inputs of ammonia, natural gas and DeNOx catalyst materials, as well as minor emissions of ammonia to air.

[10] In the 2007 version of biowaste incineration inventories in ecoinvent v2 such allocation with an economical key was already employed. In other waste incineration datasets of ecoinvent however allocation /according to motivation/ is employed and 100% of the burden is placed on the waste disposal function (Doka, G. 2007, 21ff). The energy output from average waste incineration is not burdened. Strictly speaking, the datasets from this study are not compatible with other waste

The average disposal fee for Swiss municipal waste incinerators in 2006 was 0.184 Swiss Francs (CHF) per kilogram waste (BAFU 2008: 102ff.). Average revenues from sold electricity are 0.066 CHF/kWh (Dettli, R. *et al.* 2004, Fig. 4). For sold heat from MSWI an average value of 0.0175 CHF per kWh is found (0.004861 CHF/MJ). This is for direct heat from MSWI only. Some MSWI heat is distributed in district heating systems, which need additional (fossil) heat input during winter months. According to (Dettli, R. *et al.* 2004), these combined systems have larger prices above 3 Rappen per kWh, and they were excluded in the above figure which is the market revenue for heat from MSWI. The following amounts are inventoried as output product prices, i.e. properties of intermediate exchanges.

Table 27:   Applied revenue factors for MSWI

Revenue	Unit	Amount
Disposal fee	CHF/kg	0.184
Electricity price	CHF/kWh	0.066
Electricity heat	CHF/MJ	0.004861 (0.0175 CHF/kWh)

With these values the burdens of incineration of average municipal waste with a heating value of 12 MJ/kg, would be allocated 82.88% to the disposal function, 11.08% to electricity and 6.03% to heat production. For biowaste with 4.3 MJ/kg (65% water) the allocation keys would be 93.96% on disposal, 3.64% on electricity, and 2.40% on heat.

3.5.8. Waste-specific combustion air and flue gas volume

The new MSWI model includes a calculation of the combustion air input and the resulting raw flue gas volume. Combustion air input is calculated from oxygen demand to incinerate a waste with a particular composition. Oxidation to gaseous species is considered[11] as well as oxidation of ash particles. Partial oxidation of bulk metallic iron and aluminium in bottom ash is heeded as well. From the minimal necessary, stochiometric oxygen demand the real oxygen demand is calculated using an excess air number (lambda λ) of 2, which is typical for MSWI. Total combustion air volume is calculated heeding 21 V-% oxygen in input air and an ideal gas volume R of 22.413 m^3/kmol. For an average municipal solid waste mixture a combustion air input of 6.75 m^3 and a raw flue gas volume of 7.48 m^3 results. Only a very minor part (0.4%) is due to oxidation of ashes. For other waste fractions, e.g. tin sheet packaging, the oxidation of ashes can become relevant.

3.6.   **Results**

3.6.1. Conversion processes

Below we show results for the production of unconventional crude oil from Canadian oil sands and the combustion of energy carriers used only in CHP plants (salix and miscanthus). All other energy carriers can be used in transport and the results for these are given together in Module 3.

Figure 24 shows the GHG emissions for the production of SCO from Canadian oil sands. This is compared with the results from crude oil production in other global regions. Significant variation in the GHG emissions of the conventional crude oil sources can be seen and which is a relevant factor when determining the environmental burdens imposed by the Swiss supply mix. According to ecoinvent (Jungbluth, N. 2007), 55% of the crude oil comes from Africa (not including Nigeria) and 38% from Nigeria. The higher level of emissions for crude

---

incineration datasets of the ecoinvent database, because the allocation choices are dissimilar. Numerically the differences will be relatively small.

[11]   For gaseous halogens not oxidation, but conversion to the *halogen acids* (HCl, HF etc.) is considered. This requires hydrogen from the waste matrix, which is subtracted from the hydrogen being oxidised to gaseous H_2O. So presence of halogens actually *reduces* oxygen demand, combustion air input, and flue gas output. For example 1 kg of polyethylene requires 20 m^3 of combustion air, while 1 kg of polyvinylchloride (47.5w-% Cl) needs only 10.7 m^3 air.

oil from Nigeria is mainly due to the methane which is also brought to the surface. In Nigeria this gas is flared (producing CO_2) rather than being captured and utilised as an energy resource. It is important to bear in mind that the particular SCO production route analysed here represents a modern and relatively efficient process, such that the results for other processes or extraction projects may be significantly higher.

GHG emissions of crude oil (CO) supplied to CH and Europe
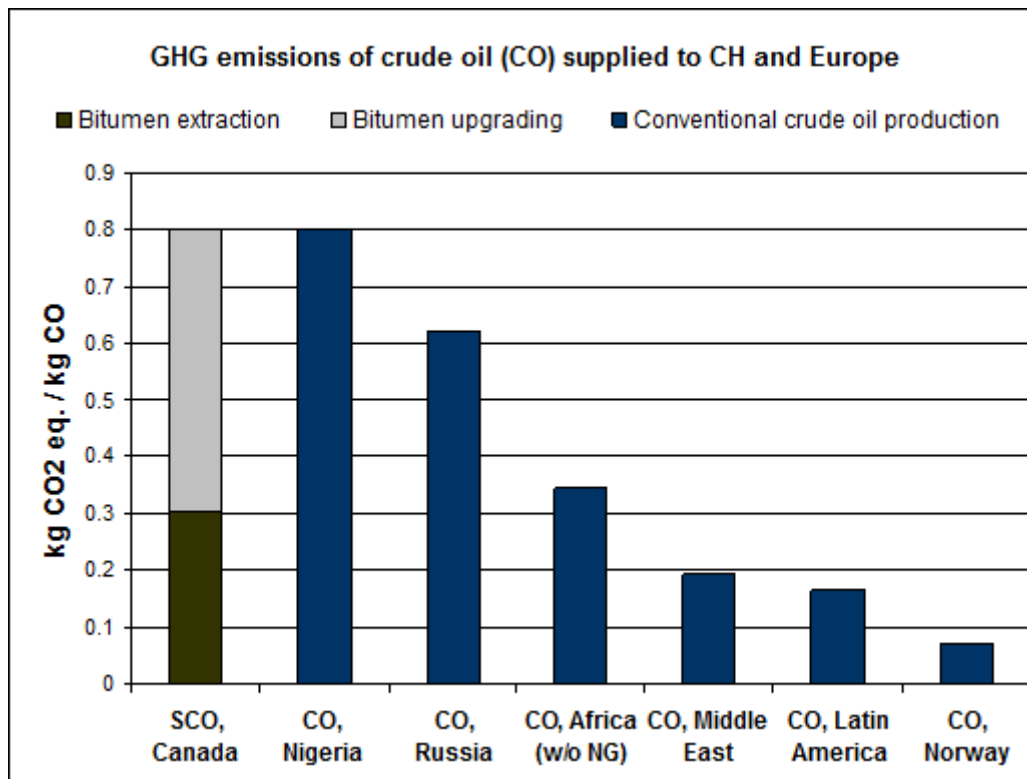
SCO: Synthetic crude oil; CO: Crude oil

Figure 24: **Greenhouse gas (GHG) emissions from the production of crude and synthetic crude oil. Data for all crude oil pathways were taken from ecoinvent (Jungbluth, N. 2007)**

The aggregated and normalized potential impacts from using new agricultural energy crops in CHP plants are shown in Figure 25. Here the results are differentiated between the upstream growing and harvesting of the fuel and the combustion in the CHP plant. In terms of per kWh electricity from the CHP plant, it can be seen that the upstream processes for salix have a higher potential impact than that of miscanthus but that the combustion of salix causes significantly less burdens. The combustion of miscanthus requires a more complex firing infrastructure and leads to higher levels of emissions and higher quantities of ash needing to be disposed of.
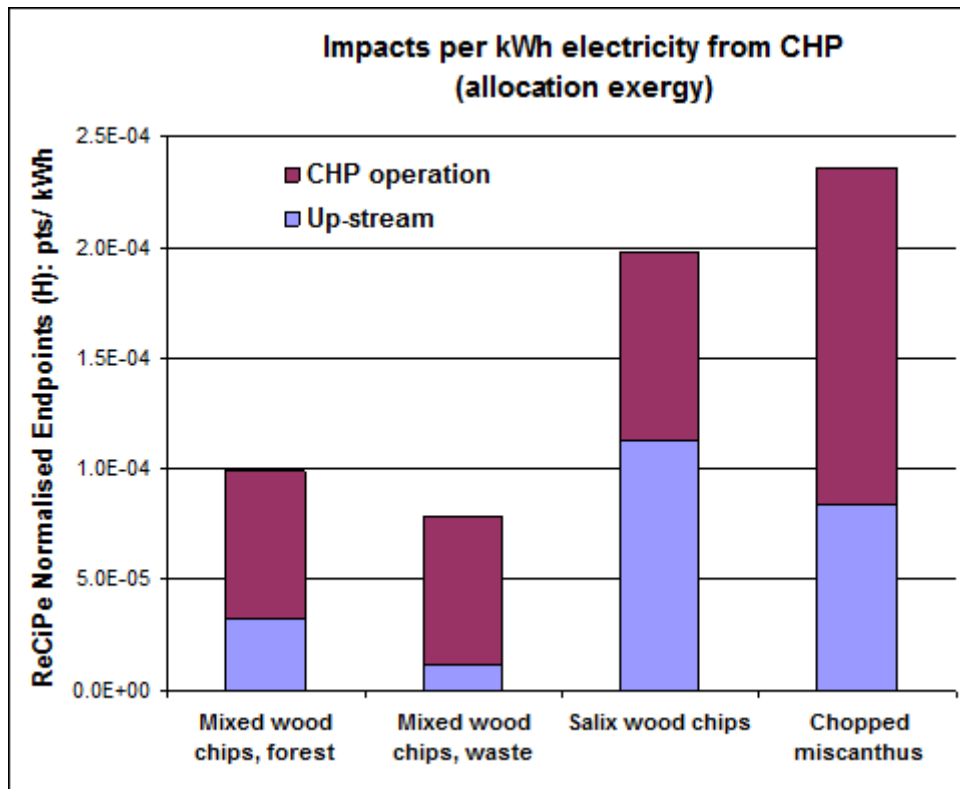
Figure 25: **Normalised aggregated impacts of combined heat and power per kWh electricity generated and using the ReCiPe methodology (Goedkoop, M. *et al.* 2009).**

The use of salix and miscanthus have been compared to the existing wood-fired CHP process (Mixed wood chips from industry waste) (Heck, T. 2007). For the purposes of generating a more relevant comparison, a process chain for the combustion of mixed wood chips from forest sources was generated and which therefore represents a similar fuel from a non-waste process stream. This used the same combustion process as for the waste wood chips from industry but with a different supply chain. Similarly to the salix wood chips, forest sourced wood chips require drying prior to combustion and here we used the same electricity demand of 10kWh/m^3 but with an additional demand for periodical mechanical turning of the woodchip pile due to the higher moisture content of the forest wood chips. This is done using a diesel fuelled machine and for this we assumed 10MJ/ m^3. The effect of these energy demands on the results were found to be insignificant at these levels, and if both are doubled then the overall results increase by 6%.

3.6.2. MSWI results

3.6.2.1. Inventory results

Table 28: Excerpt of raw process inventory results (unallocated)

All figures per kg wet waste input		biowaste, 60% water	biowaste, 65% water	biowaste, 72% water
Direct Stack Emissions				
Carbon Dioxide CO_2 (biogenic)	mg	586'600	513'200	410'500
Methane CH4 (biogenic)	mg	8.981	7.858	6.286
Nitrogen Oxides NOx	mg	200.6	175.5	140.4
Sulfur Oxides SOx	mg	6.367	5.571	4.457
Particulates PM<2.5	mg	7.5	7.649	7.856
Dioxins	ng	0.07538	0.07687	0.07896
Net Energy				
net electricity	kWh	0.1411	0.108	0.06164
net useful heat	MJ	1.189	0.9666	0.6549
waste heat emissions	MJ	6.742	5.969	4.886
Processing				
Sodium hydroxide input to flue gas treatment	mg	1242	1086	869.2
cement input for ash solidification	mg	7546	6603	5282
landfilled solids	mg	152'400	133'350	106'680
Short-term emission to water				
Arsenic As	mg	0.8988	0.7864	0.6292
Chromium Cr	mg	0.2598	0.2274	0.1819
Selenium Se	mg	0.1623	0.142	0.1136
Long-term emission to water				
Arsenic As	mg	1.095	0.958	0.7664
Chromium Cr	mg	0.8442	0.7387	0.5909
Selenium Se	mg	0.335	0.2931	0.2345
Allocation factors				
burden allocated to disposal service	%	92.42%	93.96%	96.21%
burden allocated to electricity generation	%	4.68%	3.64%	2.13%
burden allocated to heat generation	%	2.90%	2.40%	1.66%

It can generally be observed that with higher water content in the biowaste, lower emissions and expenditures result. With higher water content obviously a lower heating value of the waste results and thus less net energy output. The process burdens are allocated between disposal service and the two energy supplies, according to generated revenue. With higher water content less usable energy is produced and hence a lower percentage of the total burden is allocated to energy production.

The following table shows the allocated inventory results per functional unit (kg for disposal, kWh for electricity and MJ for heat).

Table 29: Excerpt of raw process inventory results (allocated)

Direct Stack Emissions		biowaste, 60% water			biowaste, 65% water			biowaste, 72% water		
		burden on disposal service per kg in	burden on electricity production per kWh out	burden on heat production per MJ out	burden on disposal service per kg in	burden on electricity production per kWh out	burden on heat production per MJ out	burden on disposal service per kg in	burden on electricity production per kWh out	burden on heat production per MJ out
Carbon Dioxide CO_2 (biogenic)	mg	542'100	194'500	14'320	482'200	173'000	12'740	394'900	141'700	10'430
Methane CH_4 (biogenic)	mg	8.3	2.977	0.2193	7.383	2.648	0.1951	6.048	2.169	0.1598
Nitrogen Oxides NOx	mg	185.4	66.49	4.897	164.9	59.15	4.356	135.1	48.45	3.568
Sulfur Oxides SOx	mg	5.884	2.111	0.1554	5.235	1.878	0.1383	4.288	1.538	0.1133
Particulates PM<2.5	mg	6.931	2.486	0.1831	7.187	2.578	0.1899	7.559	2.711	0.1997
Dioxins	ng	0.06966	0.02499	0.00184	0.07223	0.02591	0.001908	0.07597	0.02725	0.002007
Net Energy										
net electricity	kWh	0.1411	0	0	0.108	0	0	0.06164	0	0
net useful heat	MJ	1.189	0	0	0.9666	0	0	0.6549	0	0
waste heat emissions	MJ	6.231	2.235	0.1646	5.608	2.012	0.1482	4.7	1.686	0.12417
Processing										
Sodium hydroxide input to flue gas treatment	mg	1148	411.6	30.32	1021	366.2	26.97	836.2	299.9	22.09
cement input for ash solidification	mg	6974	2501	184.2	6204	2225	163.9	5082	1823	134.3
landfilled solids	mg	140'850	50'522	3'721	125'300	44'944	3'310	102'640	36'816	2'712
Short-term emissions to water										
Arsenic As	mg	0.8306	0.2979	0.02194	0.7389	0.2651	0.01952	0.6053	0.2171	0.01599
Chromium Cr	mg	0.2401	0.08613	0.006344	0.2136	0.07663	0.005644	0.175	0.06277	0.004623
Selenium Se	mg	0.15	0.05379	0.003962	0.1334	0.04785	0.003524	0.1093	0.0392	0.002887
Long-term emissions to water										
Arsenic As	mg	1.012	0.3629	0.02673	0.9001	0.3229	0.02378	0.7373	0.2645	0.01948
Chromium Cr	mg	0.7802	0.2798	0.02061	0.6941	0.249	0.01834	0.5685	0.2039	0.01502
Selenium Se	mg	0.3096	0.111	0.008178	0.2754	0.09878	0.007275	0.2256	0.08091	0.005959

It can be observed that the emissions and expenditures per functional unit generally decrease with increasing water content. For instance the direct NOx emissions are 66.5, 59.2, and 48.4 mg/kWh respectively for incineration of biomass with 60, 65%, and 72% water content.

So the higher the water content of the waste is the lower the burdens per generated unit of energy are. This might be a surprising finding. It means that an unsuitable fuel with high water content can have lower burdens than a suitable dry fuel, while expectations might be opposite, that a "good" fuel will be more efficient and therefore less burdening. The observed outcome is the result of the applied standard allocation procedure. The allocation procedure attributes a percentage of the total process emissions to the generated electricity or heat. To calculate the allocated emissions per kWh of electricity generated following calculation is performed (analogous for heat):

$$E_e = E_t \cdot \frac{a_e}{O_e}$$

where

E_e = Emission allocated to electricity production, per kWh electricity output

E_t = Total emission of the process, per kg waste input

a_e = allocation factor for electricity production, in % of revenue

O_e = Output of electricity, in kWh per kg waste input

As we have already seen the total process emissions decrease with water content, as does the allocation factors and the output of electricity. Both a_e and O_e are roughly proportional to $(T - w)$ where T is some systemic threshold and w is the water content of the waste. It follows that the last term (a_e/O_e) is roughly constant, and therefore E_e is roughly proportional to E_t, which we have already seen decreases with larger water content.

Of course there is a discontinuity at water contents above T, when the biowaste does not produce any net energy at all. In the region below a water content of T, the trend is decreasing burdens with increasing water content.

3.6.2.2. LCIA results

If the cumulated life cycle emissions are assessed with the ReCiPe'08(HA) LCIA method then 0.0009 – 0.0011 burden points per kWh electricity result and 0.0000644 – 0.0000841 burden points per MJ useful heat. This is below most other energy supply technologies, except for hydropower which has 0.00045 burden points per kWh electricity.

3.6.2.3. Comparison with 2007 results

In (Doka, G. 2007) similar inventories for biowaste incineration were created. Figure 26 shows a comparison of the unallocated cumulated process burdens with the results of this study. "Infrastructure" is for the MSWI plant itself and the required landfills. "Direct air emissions" are from the MSWI, while "direct water emissions" are from MSWI and short-term emissions (<100 yr) of the landfilling of ashes. "Direct LT water emissions" are long term water emissions (101-60'000yr) from ash landfills. "Auxiliaries" are for instance lime and NaOH used in the scrubbers, or cement for flue ash solidification.
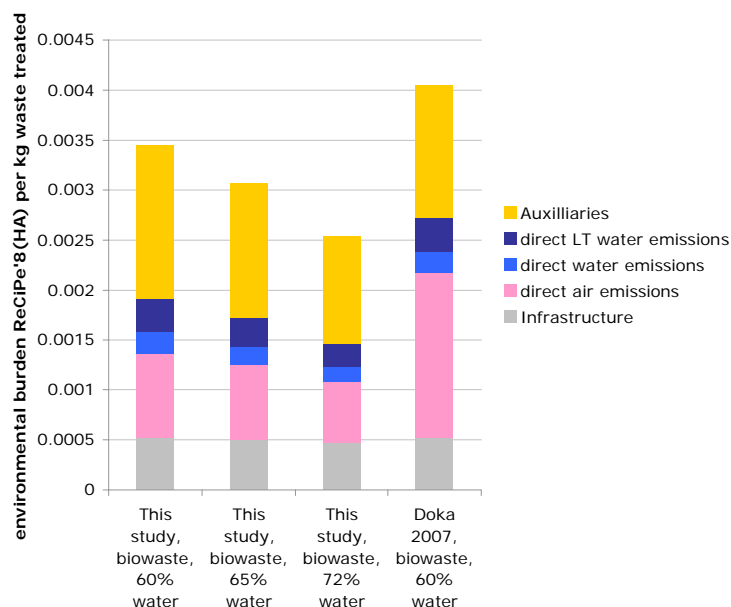
Figure 26: Contributions to unallocated MSWI process burdens per kilogram waste incinerated (LCIA with ReCiPE'08 HA)

The already mentioned decrease of burdens with increasing water content is apparent in this figure in columns 1 to 3. Comparing column 4 and column 1 includes the advancement of flue gas treatment in Swiss MSWI between 2000 and 2008, although part of that reduction is compensated by a new model to attribute emission of incomplete combustion products.

It can also be seen that auxiliaries, i.e. inputs of scrubber chemicals, cement for solidification etc. make up a significant part of the total burdens for biowaste incineration.

4. Overall evaluation

4.1. Introduction

The extension and update of the ecoinvent inventories for bioenergy allows an update of the bioenergy LCA study (Zah, R. *et al.* 2007). Here we present the environmental assessment of the new crop inventories and conversion processes for biofuel production. The results are then compared with those from the 2007 study of Zah et al. (2007).

Furthermore, important aspects of the assessment of bioenergy were investigated in this project: uncertainty analysis and the influence of the allocation method; the later will be addressed in a separate report after the release of ecoinvent v3.

4.2. Method

In this chapter we present the most important methodological aspects of this study.

4.2.1. Functional unit

The vehicle fuels are compared for one kilometre driven in an average middle weight car.

4.2.2. Temporal scope

For the new inventories, the temporal scope is 2009, while v2.2 inventories refer in general to the period 2000-2005.

4.2.3. Allocation procedure

In this study we use the allocation factors calculated for ecoinvent v2.2 for the updated inventories. For the new inventories, the same approach as in ecoinvent v2.2 was applied: if possible, flows were attributed on a physical basis (mass or energy). Where no evidence for such a relationship was available, we used economic allocation.

Allocation analysis

Ecoinvent v3 will allow calculating the results of production pathways using different allocation methods on the complete pathway, which means including background data like refineries. However, ecoinvent v3 will only be available after the end of this project. Because it is not possible to perform such a calculation and vary manually the allocation factors of the datasets in the entire database, these results will be included in a separate report containing a sensitivity analysis of the influence of the allocation methods.

Carbon balance

The attributional approach divides the flows of multi-output processes according to the corresponding allocation factor. This factor does not always correspond to physical characteristics of the co-products, which induce a bias in the mass balance of the system. For example, partitioning a multi-output process with an economic allocation approach might attribute more carbon or heavy metal emissions to a product as really flows with the co-product. To compensate this bias, ecoinvent v2.2 introduced some allocation corrections and added in the datasets where needed a virtual carbon dioxide flow which compensated the allocation factors.

The new modelling in ecoinvent v3 will provide an automatic correction of the carbon balance based on the properties of the co-products, i.e. their carbon contents (see also the quality guidelines of ecoinvent v3 (Weidema, B.P. *et al.* 2012). Consequently, no manual allocation correction is needed anymore.

4.2.4. Inventory data

The inventories of the 2007 Empa study (Zah, R. *et al.* 2007) were implemented in ecoinvent v2.01. Changes occurred in some of the feedstock inventories in the versions 2.1 and 2.2 (e.g. in emission profiles of the soybean inventories or in the US agricultural inventories; see also (Althaus, H.-J. *et al.* 2010)). These corrections had no particular trend and influenced the results in both directions. The harmonization and updates which were performed in this study are summarized in the following table and will be implemented in ecoinvent v3. The

updates and the new inventories have been prepared according to the ecoinvent v3 guidelines.

Table 30: Summary of the changes in the inventory data of bioenergy compared to ecoinvent version 2.2.

Life cycle stage	Inventories	Change made	See chapter
Cultivation	All	Harmonization of N-emission calculations Update of IPCC emission factors	Agricultural processes
Cultivation	Palm fruit MY, soybean BR, sugarcane BR	New LUC calculations	Agricultural processes
Cultivation	Palm fruit CO, sugarcane CO, alfalfa	New crops added	Agricultural processes
Cultivation	Sugarcane BR, sweet sorghum CN	Correction of heavy metal emissions in cultivation to consider ashes of bagasse spread on field	Chapter 4.2.4
Cultivation	Sugar beet, CH	Correction of heavy metal emissions in cultivation because of a too high uptake of heavy metals in v2.2.	-
Fossil oil production	Oil sand	New inventories	Conversion processes
Processing	Methane pathways, Jatropha biodiesel	New inventories	Conversion processes
Processing	CHP	Update of inventories	Conversion processes
Operation	All inventories	Update of EURO 3 inventories (update of consumption and emission profile)	Conversion processes

The following table summarizes how the inventories were combined in the comparison. For comparability reasons, the operation was calculated with 2012 inventories, so that the changes in the corrected operation dataset do not influence the v2.2 results. .Indeed, the updated operation datasets have a higher fuel consumption, which influences the whole value chain. The procedure chosen allows separating this effect from the other updates.

Table 31: Summary of the combination of the inventories for comparison purposes.

Life cycle stage	Empa study 2007	This study
Cultivation	Inventories v2.01	Inventories 2012
Processing	Inventories v2.01	Inventories v2.2, except for methane inventories (2012)
Transport	Inventories v2.01	Inventories v2.2
Operation	Inventories (Zah, R. *et al.* 2007) (EURO 3)	Inventories 2012 (EURO 3)

Correction of heavy metal emissions in cultivation

Some heavy metal emissions in agriculture were considered only partially or with standard data in ecoinvent v2. For example, the ashes of burnt bagasse in Brazil and in Columbia are

spread on the field in sugar cane cultivation; in Columbia, ashes are mixed with compost. In ecoinvent v2.2, the disposal of ashes from bagasse combustion in the ethanol process in Brazil was assessed with standard data for the disposal of wood ashes, assuming 50% disposal in sanitary landfill, 25% in landfarming and in municipal incineration; the cultivation dataset includes the ash spreading but without taking into account heavy metal emissions. However, some publications warned against the use of sugarcane bagasse ashes as a fertilizer because of their heavy metal content (Sales, A. *et al.* 2010). Here we assume that the uptake of heavy metal through the sugarcane is compensated by the ash spreading. We use the same assumption for sweet sorghum bagasse CN. To avoid double accounting we delete the disposal of ashes in the ethanol production.

Residues from palm fruit milling are partly used as fertilizers, partly combusted for the milling process. Here also the combustion was treated in ecoinvent v2.2 with an adapted wood combustion dataset. In this case we also assume in the present study that all ashes are spread finally on the fields and that in the end the uptake is compensated by this disposal. We make the same assumption for the jatropha pathway, as this crop is toxic for animals and cannot be used as a feed. To avoid double accounting, similarly to the ethanol datasets, we delete the disposal of ashes in the biodiesel production.

In the following table we summarize the assumptions concerning the fate of heavy metal in all biofuel pathways.

Table 32:   Summary of the assumptions concerning the fate of heavy metal in all biofuel pathways.

Fuel	Process with allocation	Co-product	Goes to / is used as	Remark (modelling in ecoinvent v2.2)	Assumption for modelling
ETOH, sugar beet (CH)	Fermentation	Wet pulp, or beet chips	Livestock feed	Heavy metal from manure use in sugar beet cultivation is taken into account	Total heavy metal content in plant goes into livestock system
	Fermentation	Vinasse	Fertilizer	No info about heavy metals. Heavy metal content should be very low (Madejón, E. *et al.* 2001)	
ETOH, sugar cane (BR/CO)	Crushing of cane	Bagasse	Combustion	Heavy metal emissions of combustion are taken into account with standard figures. No emissions from ashes into soil are considered.	Total heavy metal content in plant goes back to field (ash disposal deleted)
	Distillation	Stillage (vinasse)	Fertilizer	No data on heavy metal content of stillage. Stillage should however have a very low heavy metal content; no effect on heavy metal content of soil by regular application (Soobadar, A. *et al.* 2011)	
ETOH, rye (RER) / corn, wheat (US)	Fermentation & distillation	DDGS	Livestock feed	No data on heavy metal content of DDGS	Total heavy metal content in plant goes into livestock system
	Dehydration	liquid effluents	Wastewater treatment plant	No data on heavy metal content of these "liquid effluents"	
ETOH, sweet sorghum (CN)	Juice extraction	Bagasse	Combustion	No data on heavy metal content of bagasse. No information on use of bagasse ashes. However, process very similar to sugarcane.	Total heavy metal content in plant goes back to field
	Fermentation & distillation	Vinasse	Fertilizer/or randomly disposed of	No data on heavy metal content of vinasse. However, process very similar to sugarcane.	
BD, rapeseed (CH/RER)	Milling	Meal	Livestock feed	Heavy metal from manure use in rapeseed cultivation is taken into account	Total heavy metal content in plant goes into livestock system
BD, Soybean (BR/US)	Milling	Meal	Livestock feed	Heavy metal from manure use in cultivation is taken into account	Total heavy metal content in plant goes into (often foreign) livestock system
BD, oil palm (MY/CO)	Milling	Palm kernel meal	Livestock feed	Heavy metal from manure use in cultivation is taken into account	Heavy metal content in kernel goes into livestock system.
		Shell, fibres	Combustion	Heavy metal in ashes not considered. Emissions in air are calculated with values for wood.	Total heavy metal in plant back to field (ash disposal deleted)
BD, jatropha (IN/AFR)	Milling	Meal	Fertilizer	Jatropha meal is toxic → use as feed is not possible	Total heavy metal content in plant goes back to field
Biogas, alfalfa	Digestion in refinery	Fibre, proteins	Diverse		Total heavy metal content in plant goes into co-product system

4.2.5. Impact indicators

In the following table we summarize the midpoint and endpoint methods used in this study.

Midpoint indicators allow a detailed evaluation of the environment impacts by focussing on specific effects on the environment. It is a problem-oriented approach which translates impacts into environmental themes such as climate change, acidification, human toxicity, etc. Such results on midpoint level can show trade-offs between the different impacts. Therefore an aggregation of the results on endpoint level is an important complement of such an analysis.

Endpoint impact categories, also known as the damage-oriented approach, translate environmental impacts into issues of concern such as human health, natural environment, and natural resources. The motivation to calculate the endpoint indicators is that the large number of midpoint indicators is very difficult to interpret, partially as there are too many, partially because they have a very abstract meaning. How to compare radiative forcing with base saturation numbers that express acidification? The indicators at the endpoint level are intended to facilitate easier interpretation, as there are only three, and they have a more understandable meaning[12].

Method	Rationale
Midpoints indicators	
Global Warming Potential (GWP) 100a IPCC 2007	The use of biofuels and bioenergy is motivated by their potential for greenhouse gas emissions mitigation (IPCC 2007).
Global Warming Potential (GWP) 100a IPCC 2001	This indicator is used for comparison with the 2007 study. (IPCC 2001)
ILCD Midpoints	Selected Midpoint indicators following the recommendations of the International Reference Life Cycle Data System (ILCD) Handbook (EC-JRC-IES 2012).
CML (characterisation)	This midpoint method is used for comparison with the 2007 study (CML 2001)
Endpoint indicators	
Eco-indicator (EI) '99 single scope H/A	This endpoint method is used for comparison with the 2007 study (Goedkoop, M. *et al.* 2001)
Swiss Ecological Scarcity method (UBP), single score v1.06	This endpoint method is used for comparison with the 2007 study as well as for the new value chains because of its Swiss focus (Frischknecht, R. *et al.* 2008).
Recipe Endpoints, World normalization factors, HA, v1.07	ReCiPe is the most actual European method (Goedkoop, M. *et al.* 2009) and has merged the midpoint approach of CML and endpoint approach of Eco-indicator
Recipe Endpoints, European normalization factors, HA, v1.07	This method is used as a sensitivity analysis to evaluate the effect of the normalization factors on the endpoint results (Goedkoop, M. *et al.* 2009)

The ILCD 2011 Midpoint method (EC-JRC-IES 2012) was released by the European Commission, Joint Research Centre in 2012. It supports the correct use of the characterisation factors for impact assessment as recommended in the ILCD guidance document "Recommendations for Life Cycle Impact Assessment in the European context -

---

[12] http://www.lcia-recipe.net/

based on existing environmental impact assessment models and factors" (EC-JRC-IES 2011).

This LCIA method includes 16 midpoint impact categories:

1. **Climate change**: Global Warming Potential calculating the radiative forcing over a time horizon of 100 years. (IPCC 2007).

2. **Ozone depletion**: Ozone Depletion Potential (ODP) calculating the destructive effects on the stratospheric ozone layer over a time horizon of 100 years. | World Meteorological Organization (WMO 1999).

3. **Human toxicity, cancer effects**: Comparative Toxic Unit for human (CTUh) expressing the estimated increase in morbidity in the total human population per unit mass of a chemical emitted (cases per kilogramme). (Rosenbaum, R.K. *et al.* 2008)

4. **Human toxicity, non-cancer effects**: Comparative Toxic Unit for human (CTUh) expressing the estimated increase in morbidity in the total human population per unit mass of a chemical emitted (cases per kilogramme). (Rosenbaum, R.K. *et al.* 2008)

5. **Particulate matter**: Quantification of the impact of premature death or disability that particulates/respiratory inorganics have on the population, in comparison to PM2.5. It includes the assessment of primary (PM10 and PM2.5) and secondary PM (incl. creation of secondary PM due to Sox, NOx and NH3 emissions) and CO. (Rabl, A. *et al.* 2004)

6. **Ionizing radiation HH** (human health): Quantification of the impact of ionizing radiation on the population, in comparison to Uranium 235. (Frischknecht, R. *et al.* 2000)

7. **Ionizing radiation E** (ecosystems) [note: this method is classified as interim; see reference for explanation]: Comparative Toxic Unit for ecosystems (CTUe) expressing an estimate of the potentially affected fraction of species (PAF) integrated over time and volume per unit mass of a radionucleide emitted (PAF m3 year/kg). Fate of radionucleide based on USEtox consensus model (multimedia model). Relevant for freshwater ecosystems. (Garnier-Laplace, J.C. *et al.* 2008)

8. **Photochemical ozone formation**: Expression of the potential contribution to photochemical ozone formation. Only for Europe. It includes spatial differentiation (Van Zelm, R. *et al.* 2008)

9. **Acidification**: Accumulated Exceedance (AE) characterizing the change in critical load exceedance of the sensitive area in terrestrial and main freshwater ecosystems, to which acidifying substances deposit. European-country dependent. (Seppälä, J. *et al.* 2006) and (Posch, M. *et al.* 2008).

10. **Terrestrial eutrophication**: Accumulated Exceedance (AE) characterizing the change in critical load exceedance of the sensitive area, to which eutrophying substances deposit. European-country dependent. (Seppälä, J. *et al.* 2006) and (Posch, M. *et al.* 2008).

11. **Freshwater eutrophication**: Expression of the degree to which the emitted nutrients reaches the freshwater end compartment (phosphorus considered as limiting factor in freshwater). European validity. Averaged characterization factors from country dependent characterization factors. (Goedkoop, M. *et al.* 2009)

12. **Marine eutrophication**: Expression of the degree to which the emitted nutrients reaches the marine end compartment (nitrogen considered as limiting factor in marine water). European validity. Averaged characterization factors from country dependent characterization factors. (Goedkoop, M. *et al.* 2009)

13. **Freshwater ecotoxicity**: Comparative Toxic Unit for ecosystems (CTUe) expressing an estimate of the potentially affected fraction of species (PAF) integrated over time and volume per unit mass of a chemical emitted (PAF m3 year/kg). Specific groups of chemicals requires further works. (Rosenbaum, R.K. *et al.* 2008)

14. **Land use**: Soil Organic Matter (SOM) based on changes in SOM, measured in (kg C/m2/a). Biodiversity impacts not covered by the data set. (Milà i Canals, L. *et al.* 2007)

15. **Water resource depletion**: Freshwater scarcity: Scarcity-adjusted amount of water used. (Frischknecht, R. *et al.* 2008).

16. **Mineral, fossil & renewable resource depletion**: Scarcity of mineral resource with the scarcity calculated as 'Reserve base'. It refers ro identified resource that meets specified minimum physical and chemical criteria related to current mining practice. The reserve base may encompass those parts of the resources that have a reasonable potential for becoming economically available within planning horizons beyond those that assume proven technology and current economics. (Van Oers, L. *et al.* 2002)

4.2.6. Uncertainty analysis

We use Monte Carlo analysis for assessing the uncertainty of the results. To allow the Monte Carlo simulation, we separate the indicator land use transformation in two categories, land use transformation from and land use transformation to, because these two parameters are not independent, which is a requirement for the Monte Carlo calculations.

4.2.7. Limitations of the study

Consistency of the results of this study with ecoinvent v3

The modelling in ecoinvent version 3 presents new features and differs fundamentally from the earlier versions. Unfortunately, due to delays related to this important update of the database, it was not possible to implement the original plan in this project, namely to use ecoinvent version 3 for the calculations. Instead, the calculations were done with Simapro and the version 2 modelling approach. The results of the same calculations with ecoinvent version 3 might therefore differ from the results presented here.

Further limitations

Due to the limited resources, it was not possible to update all flows of the ecoinvent v2.2 agricultural inventories (e.g. harmonization of phosphate emissions in a similar procedure as for nitrogen), as well as all conversion processes. The new transport inventories for EURO4/ EURO5 vehicles are being calculated currently and will be included in ecoinvent later this year. Despite these limitations, the results presented here represent a big step for the improvement of the bioenergy database.

4.3. Overall assessment

4.3.1. Midpoint indicators (ILCD 2011 Midpoints)

The results of the selected Midpoint indicators according to ILCD show distinctly that for many of the indicators, the biofuel value chains have higher impact values than gasoline low-sulphur (as a fossil reference), except for the methane pathways. Only climate change and ozone depletion are more favourable in most of the biofuel pathways than the fossil reference. Higher values for biofuel production pathways are found in indicators relevant for agricultural processes (eutrophication, acidification, water depletion, ecotoxicity, land use).

Table 33: IPCC (Climate change) and ILCD Midpoint results of the biofuel and fossil fuel value chains per v.km.

	Climate change	Freshwater eutrophication	Terrestrial eutrophication	Marine eutrophication	Acidification	Ozone depletion	Photochemical ozone formation	Ionizing radiation ecosystem (interim)	Freshwater ecotoxicity	Human toxicity, cancer effects	Human toxicity, non-cancer effects	Particulate matter	Ionizing radiation, human health	Water resource depletion	Land use	Mineral, fossil & ren resource depletion
	kg CO2 eq	kg P eq	molc N eq	kg N eq	molc H+ eq	kg CFC-11 eq	kg NMVOC eq	CTUe	CTUe	CTUh	CTUh	kg PM2.5 eq	kg U235 eq	m3 water eq	kg C deficit	kg Sb eq
Rape seed ME IP/CH	2.1E-01	7.7E-05	1.3E-02	3.1E-03	3.2E-03	2.2E-08	1.0E-03	2.3E-07	7.7E-01	2.2E-08	5.7E-07	2.0E-04	7.7E-02	1.1E-04	4.9E+01	7.0E-06
Rape seed ME EXT/CH	2.1E-01	6.4E-05	1.9E-02	3.6E-03	4.5E-03	2.4E-08	1.0E-03	2.3E-07	7.1E-01	2.0E-08	6.6E-07	2.2E-04	7.7E-02	1.1E-04	5.6E+01	7.0E-06
Rape seed ME conv/DE	1.7E-01	7.5E-05	4.7E-03	7.2E-04	1.3E-03	1.6E-08	9.6E-04	2.0E-07	4.1E-01	2.2E-08	2.3E-08	1.5E-04	6.6E-02	1.1E-04	3.6E+00	6.9E-06
Rape seed ME conv/FR	2.1E-01	1.0E-04	1.1E-02	2.2E-03	2.9E-03	1.8E-08	9.8E-04	2.0E-07	1.1E+00	2.4E-08	6.7E-08	2.0E-04	6.7E-02	1.1E-04	4.5E+00	7.0E-06
Soybean ME BR	8.4E-01	6.6E-05	4.1E-03	1.8E-03	1.3E-03	1.3E-08	1.6E-03	1.9E-07	5.3E+00	2.2E-08	-1.8E-07	2.8E-04	6.2E-02	9.7E-05	1.4E+01	8.8E-05
Soybean ME US	1.4E-01	6.3E-05	3.0E-03	1.6E-03	1.0E-03	1.2E-08	1.0E-03	1.9E-07	2.3E-01	2.0E-08	-1.7E-07	1.3E-04	6.3E-02	9.6E-05	4.8E+00	6.5E-06
Jatropha ME EXT/IN	-2.2E-01	6.2E-05	1.5E-02	2.6E-03	3.6E-03	1.4E-08	1.3E-03	2.0E-07	6.3E-01	2.2E-08	5.4E-07	2.8E-04	6.6E-02	6.1E-04	-4.0E+01	5.6E-05
Jatropha ME INT/IN	1.4E-01	1.0E-04	4.7E-02	1.8E-03	1.1E-02	2.4E-08	1.3E-03	2.6E-07	9.5E-01	4.5E-08	6.5E-07	5.2E-04	8.5E-02	3.3E-03	-1.2E+01	1.7E-05
Jatropha ME fence/AFR	-1.4E-02	4.1E-05	2.3E-03	4.2E-04	7.5E-04	8.8E-09	6.9E-04	1.9E-07	2.8E-01	1.8E-08	1.8E-08	1.2E-04	6.1E-02	8.7E-05	2.9E-01	6.2E-06
Jatropha ME EXT/AFR	-7.0E-01	4.9E-05	3.4E-03	1.6E-03	1.0E-03	9.0E-09	7.7E-04	1.9E-07	4.4E-01	2.0E-08	2.4E-07	1.5E-04	6.2E-02	8.8E-05	-5.4E+01	-1.8E-05
Palm fruit ME MY	3.3E-01	5.2E-05	5.0E-03	1.5E-03	1.4E-03	1.1E-08	1.1E-03	2.2E-07	4.7E-01	2.0E-08	3.4E-08	2.9E-04	7.1E-02	3.2E-03	2.1E+00	3.5E-05
Palm fruit ME CO	8.4E-02	4.7E-05	4.2E-03	3.7E-04	1.2E-03	1.2E-08	9.9E-04	1.8E-07	5.3E-01	1.9E-08	7.7E-08	2.6E-04	5.9E-02	3.4E-04	2.0E+00	9.9E-06
Sugar cane molasses ETOH BR	1.2E-01	5.5E-05	8.3E-03	9.3E-04	2.2E-03	1.2E-08	8.8E-04	1.9E-07	8.6E-01	2.4E-08	3.6E-07	5.7E-04	6.1E-02	3.6E-04	5.4E+00	6.8E-06
Sugar cane ETOH BR	1.2E-01	5.5E-05	7.4E-03	8.1E-04	2.1E-03	1.2E-08	9.1E-04	1.9E-07	7.5E-01	2.3E-08	2.7E-07	5.3E-04	6.2E-02	5.0E-04	4.4E+00	6.9E-06
Sugar cane ETOH CO	1.3E-01	4.5E-05	8.7E-03	7.3E-04	2.2E-03	1.1E-08	1.2E-03	1.8E-07	7.1E-01	2.1E-08	1.8E-07	5.7E-04	6.1E-02	1.6E-02	3.6E-01	6.6E-06
Maize ETOH US	2.7E-01	9.8E-05	8.2E-03	1.5E-03	2.5E-03	2.7E-08	1.0E-03	2.3E-07	1.3E+00	2.3E-08	-3.7E-08	1.8E-04	8.0E-02	4.5E-04	5.5E+00	7.0E-06
Rye ETOH CONV/RER	3.1E-01	1.2E-04	1.0E-02	5.9E-03	2.9E-03	2.8E-08	1.2E-03	2.6E-07	1.5E+00	2.9E-08	6.5E-07	2.0E-04	8.6E-02	2.2E-04	1.2E+02	7.8E-06
Sugar beet ETOH IP/CH	1.2E-01	4.6E-05	3.9E-03	5.7E-04	1.2E-03	1.3E-08	5.7E-04	2.2E-07	2.5E-01	1.9E-08	-8.4E-08	1.1E-04	7.2E-02	1.2E-04	9.3E-01	6.5E-06
Sweet sorghum stem ETOH CN	1.3E-01	6.9E-05	5.8E-03	4.3E-04	1.9E-03	1.3E-08	1.0E-03	2.5E-07	7.6E-01	2.2E-08	1.1E-07	5.2E-04	8.2E-02	5.2E-03	1.5E+00	6.8E-06
Wheat ETOH US	3.4E-01	2.7E-04	1.4E-02	1.4E-03	4.0E-03	2.8E-08	1.4E-03	2.6E-07	9.4E-02	-1.1E-08	-3.7E-07	2.5E-04	8.7E-02	4.1E-03	1.7E+01	8.4E-06
Wheat ETOH CONV/DE	2.7E-01	9.6E-05	6.8E-03	1.5E-03	2.0E-03	2.5E-08	1.0E-03	2.5E-07	4.9E-01	2.4E-08	6.8E-08	1.6E-04	8.2E-02	2.0E-04	6.3E+00	7.4E-06
Wheat ETOH CONV/ES	3.5E-01	1.8E-04	8.8E-03	4.0E-03	2.6E-03	3.1E-08	1.5E-03	2.6E-07	6.1E-01	3.2E-08	3.2E-07	2.2E-04	8.7E-02	2.5E-04	1.2E+01	8.6E-06
Wheat ETOH CONV/FR	2.8E-01	1.2E-04	1.6E-02	3.8E-03	4.1E-03	2.8E-08	9.8E-04	2.4E-07	5.7E-01	2.5E-08	1.0E-07	2.3E-04	7.9E-02	2.0E-04	6.4E+00	7.4E-06
Methane 96%, Alfalfa grass	2.3E-01	7.3E-05	5.6E-03	2.1E-04	1.6E-03	1.0E-08	5.4E-04	4.0E-07	2.6E-01	2.0E-08	-6.3E-08	1.1E-04	1.3E-01	2.1E-04	1.4E+00	6.7E-06

	Climate change	Freshwater eutrophication	Terrestrial eutrophication	Marine eutrophication	Acidification	Ozone depletion	Photochemical ozone formation	Ionizing radiation ecosystem (interim)	Freshwater ecotoxicity	Human toxicity, cancer effects	Human toxicity, non-cancer effects	Particulate matter	Ionizing radiation, human health	Water resource depletion	Land use	Mineral, fossil & ren resource depletion
	kg CO2 eq	kg P eq	molc N eq	kg N eq	molc H+ eq	kg CFC-11 eq	kg NMVOC eq	CTUe	CTUe	CTUh	CTUh	kg PM2.5 eq	kg U235 eq	m3 water eq	kg C deficit	kg Sb eq
Methane 99%, sewage sludge	1.4E-01	4.6E-05	1.1E-03	8.9E-05	5.3E-04	1.6E-08	4.2E-04	3.3E-07	3.0E-01	1.9E-08	1.7E-08	8.1E-05	1.1E-01	1.4E-04	2.9E-01	6.2E-06
Methane 96%, wood chips	7.0E-02	3.9E-05	1.1E-03	8.6E-05	5.2E-04	7.5E-09	4.2E-04	2.0E-07	2.8E-01	1.8E-08	1.8E-08	8.0E-05	6.7E-02	9.1E-05	2.9E-01	6.4E-06
Methane 96%, cattle slurry	9.2E-02	4.9E-05	1.6E-03	1.4E-04	6.8E-04	9.9E-09	5.7E-04	3.0E-07	3.6E-01	2.1E-08	9.2E-08	9.4E-05	1.0E-01	1.3E-04	4.2E-01	7.4E-06
Fossil diesel, low-sulfur	2.7E-01	3.9E-05	2.2E-03	2.1E-04	8.2E-04	4.3E-08	8.2E-04	1.8E-07	2.9E-01	1.8E-08	2.0E-08	2.1E-04	5.8E-02	8.8E-05	6.0E-01	6.1E-06
Fossil gasoline, low-sulfur	3.2E-01	4.0E-05	1.7E-03	1.6E-04	8.0E-04	5.1E-08	7.4E-04	1.8E-07	2.9E-01	1.8E-08	2.0E-08	1.2E-04	6.0E-02	9.2E-05	6.0E-01	6.1E-06
Natural gas	2.6E-01	3.7E-05	1.1E-03	9.4E-05	5.7E-04	3.9E-08	4.7E-04	2.0E-07	2.8E-01	1.8E-08	1.7E-08	7.9E-05	6.5E-02	8.7E-05	3.6E-01	6.0E-06
Diesel, SCO	3.0E-01	4.0E-05	2.1E-03	2.0E-04	1.3E-03	9.6E-09	7.2E-04	1.8E-07	3.4E-01	2.1E-08	2.1E-08	2.6E-04	5.8E-02	3.9E-02	5.2E-01	6.0E-06
Gasoline, SCO	3.5E-01	4.2E-05	1.6E-03	1.5E-04	1.4E-03	1.2E-08	6.2E-04	1.8E-07	3.5E-01	2.2E-08	2.0E-08	1.8E-04	5.9E-02	4.3E-02	5.8E-01	6.0E-06

Higher than 95% of reference	Reference	Lower than 105% of reference	
5%-40% lower than reference	More than 40% lower than reference	5%-67% higher than reference	More than 67% higher than reference

The greenhouse gas emissions (*climate change* indicator) are for many biofuel pathways higher than 60% of the fossil reference, which is the threshold defined in the Swiss legislation (MinöStV 2008). Two factors are important in this regard:

- Land use change emissions (soybean BR, palm fruit MY)
- Dinitrogen monoxide (N_2O) emissions in the agricultural phase

Freshwater eutrophication is caused by the phosphate emissions as a consequence of the application of phosphor fertilisers during crop cultivation. Thus, biofuel pathways show a higher impact on freshwater eutrophication than fossil fuels, except for biofuels based on waste biomass which show similar impacts as fossil fuels.

Terrestrial eutrophication is caused by the deposition of aerial nitrogen compounds, which induces growth and competitiveness of vegetation in ecosystems limited availability of nitrogen. Such emissions stem from the application of nitrogen fertilizers during crop cultivation and in the case of nitrogen oxides also from fuel combustion. Consequently, the emissions are higher for biofuel pathways than for fossil fuels, except for biofuels based on waste biomass.

Marine eutrophication is caused by nitrogen emissions (ammonia, nitrate, nitrogen oxides) in water and air stemming from the nitrogen fertilizer application during crop cultivation. The results for this indicator are therefore very similar to those of terrestrial eutrophication.

Generally, ammonia accounts for about half of the *terrestrial acidification* impacts in the biofuel pathways. These emissions are due to the nitrogen contained in the organic and mineral fertilizers and are about 100 times higher than for the fossil pathways. Jatropha hedges are grown without fertilizer, which explains why they perform better here. The rest of the impacts are caused by nitrogen oxides and sulphur dioxides which are emitted during the fuel combustion in the same amount for biofuels and fossil fuels and as a consequence of fossil fuel combustion in the rest of the production chain.

The production of conventional crude oil emits relatively large amounts of "methane, bromotrifluoro-, Halon 1301". Therefore, the results for the *ozone depletion* indicator are proportional to the use of fossil fuel. Ozone depleting emissions are lower for petrol and diesel from SCO because the available data on oil sands mining and upgrading does not report such emissions to be as high as for conventional crude oil.

Photochemical oxidant formation impacts are mainly caused by the emissions of nitrogen oxides, which occur in agriculture as an indirect consequence of nitrogen fertilizers. Further, nitrogen is also emitted during the combustion of fuels in vehicles, machines or stationary equipment. During the combustion process NMVOCs are also released. These substances are the second most important emissions contributing to the photochemical oxidant formation.

Ionising radiation is due to the use of nuclear energy and occurs therefore only in background processes. It reflects the use of electricity from nuclear power plants and has no relevance in these systems.

(Freshwater) ecotoxic effects are mainly caused by the emissions of pesticides and heavy metals in agriculture, leading to a relatively high impact of biofuel pathways compared to fossil value chains. The ecotoxic effect of fossil fuel pathways are caused by substances such as chromium, antimony or zinc released in infrastructure processes.

Human toxicity of biofuels and fossil fuels are mainly caused by heavy metal emissions in soil as well as some pesticides like aldrin for the non-cancer toxicity, and by dioxin, formaldehyde and lead emissions in air for the cancer toxicity. These emissions occur mainly in combustion processes of the fuels (transport, machines and stationery equipment), in the use of the car as well as in agriculture for pesticides.

Particulate matter formation occurs because of combustion but also as a consequence of ammonia emissions. This explains why the biofuel pathways have high values for this indicator. Furthermore, it is assumed that 20% of the above ground biomass is burned by transformation of forest, so that the particulate formation is especially high for biofuel pathways where forest is transformed.

Irrigated crops (US wheat and maize, sweet sorghum CN, sugar cane CO, palm fruit MY, jatropha INT/IN) have the highest *water depletion* impacts; the water requirements vary

between 17 l/kg seed (extensive jatropha in India) and 110l/kg seed (intensive jatropha production). The SCO crude oil production consumes about 4'000 litres water per kg bitumen, which leads to the very high values for this production pathway. For the other biofuels, water depletion is due to the production of ethanol or to background processes like fertilizer production (production of sulphuric acid for phosphate fertilizers).

Land use accounts for the deficit in soil carbon due to land transformation and occupation. The transformation of an area in arable land is therefore causing higher values than the transformation in pasture and meadows or forest e.g. Occupation of arable land is also inducing soil carbon deficit. Therefore all agricultural activities induce such an impact; it is only very low or even negative when transformation to forest or permanent crop occurs. The characterisation factors however are not yet complete, so that this assessment must be taken cautiously. The factor for permanent crop, fruit, does not exist, which means that the transformation of arable land in this category like in the case of jatropha results in negative impacts. This results must be considered as too favourable.

The *resource depletion* indicator comprises the depletion of fossil (gas, oil, coal), metallic (all relevant metals) and other (carbon in soil, peat, phosphorus, sulphur, talc, uranium) re-sources. The depletion of carbon in soil is especially relevant for those biofuel pathways where land use change occurs (jatropha IN, palm MY, soybeans BR, sugarcane BR) and inversely the amelioration of the carbon content in soil delivers negative (and therefore fa-vourable, like in the case of extensive jatropha in Africa) results. Uranium has a relatively high characterisation factor, so that the use of nuclear electricity influences much the results, as in the Swiss methane or wheat ethanol US pathway. However, in the latter case, this re-sult must be asserted with caution, as it is partly caused by default background datasets for pesticides or agricultural machinery where the country (and therefore the electricity mix) does not coincide with the wheat dataset. On the contrary, for biogas and biofuel pathways based on Swiss feedstocks, this dominance of nuclear electricity is justified.

4.3.2. Endpoint indicators

Swiss EcoScarcity 2006 (UBP)

The UBP results of updated and new biofuel pathways follow a similar trend to the one shown in the Empa study from 2007: most biofuels have higher impacts measured with the ecological scarcity method than the fossil reference gasoline (see Figure 27). Overall, the agricultural processes account for a great proportion of the impacts, which are caused by nitrate, phosphate, pesticides and heavy metal emissions. The new accounting of heavy metal emissions from ashes (s. chapter 4.2.3) and the correction of these flows in the sugar beet cultivation compared to ecoinvent v2 leads to higher values in those pathways (palm oil, sugarcane, sweet sorghum). On the contrary, methane pathways based on waste products perform better than the fossil reference, because no impacts of agricultural processes are allocated to biowaste.
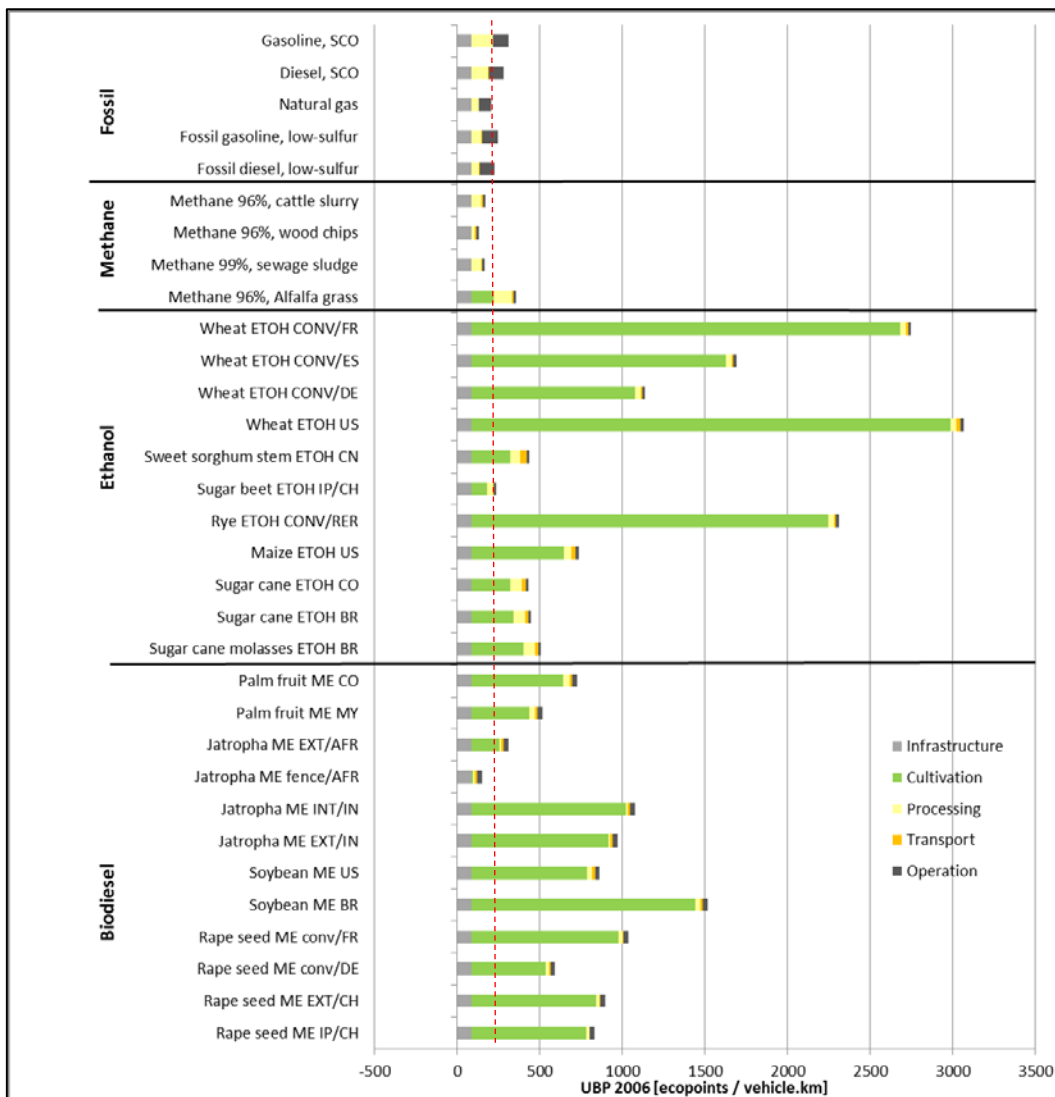
Figure 27: Results calculated with the Swiss ecological scarcity method (points per v.km), by life cycle stage.

ReCiPe World, H/A

In contrast to the results calculated with the Swiss ecological scarcity method, biofuel pathways perform in most cases better than the fossil reference when the ReCiPe World H/A method is used.

The biofuel production pathways with higher values than the fossil reference are those with high natural land transformation impacts like soybean and palm fruit methyl ester MY, which is also linked to high CO_2 emissions from land use change. Biofuels with a high fossil energy use in the conversion like some ethanol pathways have also impacts in the same range as the fossil fuels. On the other extreme, biofuels from extensive jatropha production in Africa achieve very low or even negative values. However, the production potential of this system is restricted as the yields are very low.
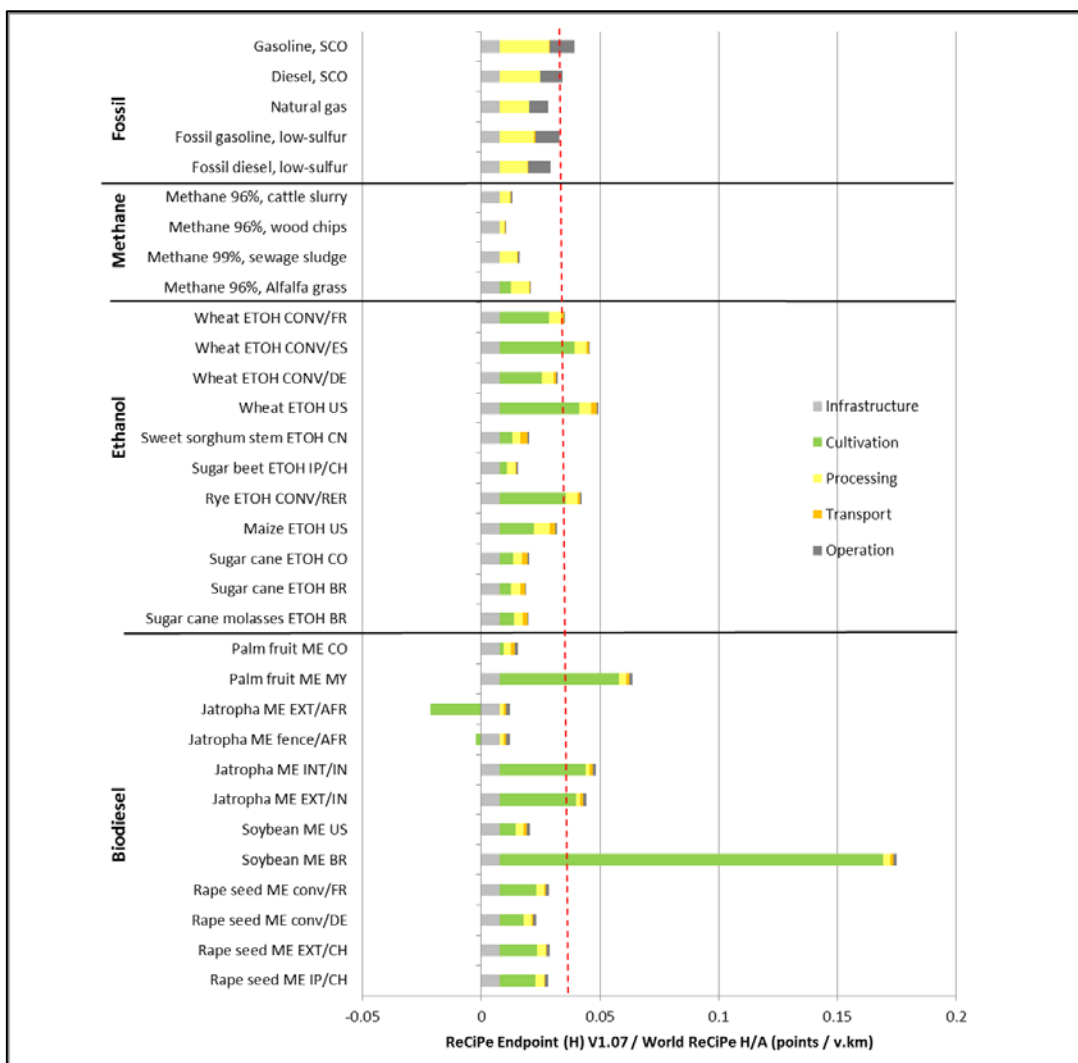
Figure 28: ReCiPe World (H/A) Endpoint results (points per v.km), by life cycle stage.

The following figure illustrates the weighting of the ReCiPe method. Here we show the results by the area of protections. The use of fossil resources as well as the emissions of greenhouse gases is responsible for most of the impacts calculated with this method. Where deforestation occurs, natural land transformation influences greatly the results (soybeans, palm fruit). Agricultural land occupation influences the results of biofuel from crops with a low yield in the field (jatropha) or a low energy conversion yield (wheat).
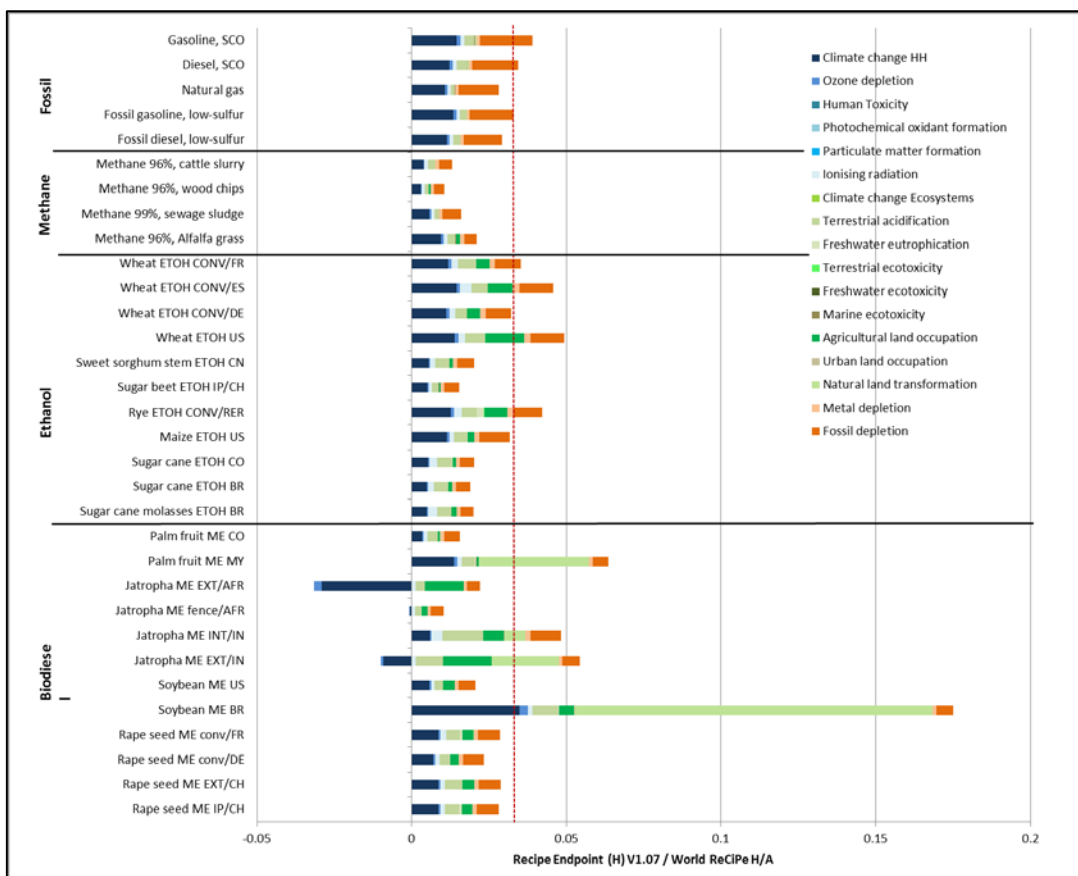
Figure 29: ReCiPe World (H/A) Endpoint results (points per v.km), by impact category.

ReCiPe Endpoint (H) V1.06 / Europe ReCiPe H/A

ReCiPe Endpoint Europe uses normalization factors for EU 25+3. The emission pattern in Europe differs from the rest of the world, with for example a very low share of the overall world ozone depletion emissions, and on the other side of the scale a high share of ionising radiation emissions (Sleeswijk, A.W. *et al.* 2008). This explains the very different results with these normalization factors (Figure 30). Indeed, assessing the fuel pathways with the ReCiPe Endpoint Europe method leads to an increase in the impact compared to the ReCiPe method with global normalization factors. Even though the ranking of the biofuels stays similar, most biofuels show ReCiPe Endpoint results higher than or nearly equal to the fossil reference gasoline, whereas with ReCiPe World almost all biofuels performed better than the reference. With the European normalization factors, Jatropha from Africa does not achieve negative values any more.
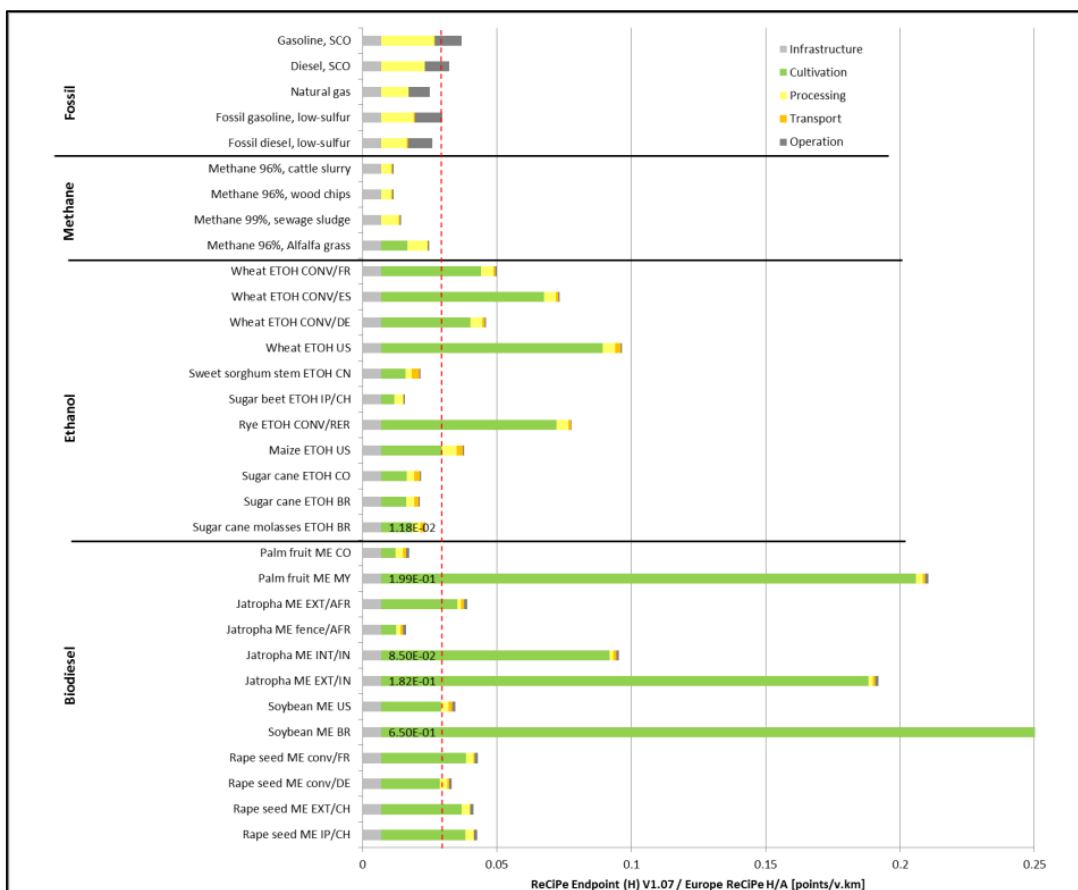
Figure 30: ReCiPe Europe (H/A) Endpoint results (points per v.km), by life cycle stage.

This difference is due to a large extent to the normalisation factors for natural land transformation, which is hundred times higher for the European factors than for the world factors, as well as to the factors for agricultural land occupation which are also about two times higher. All other normalisation factors are lower for Europe, except for ozone depletion, which does not influence much the results.
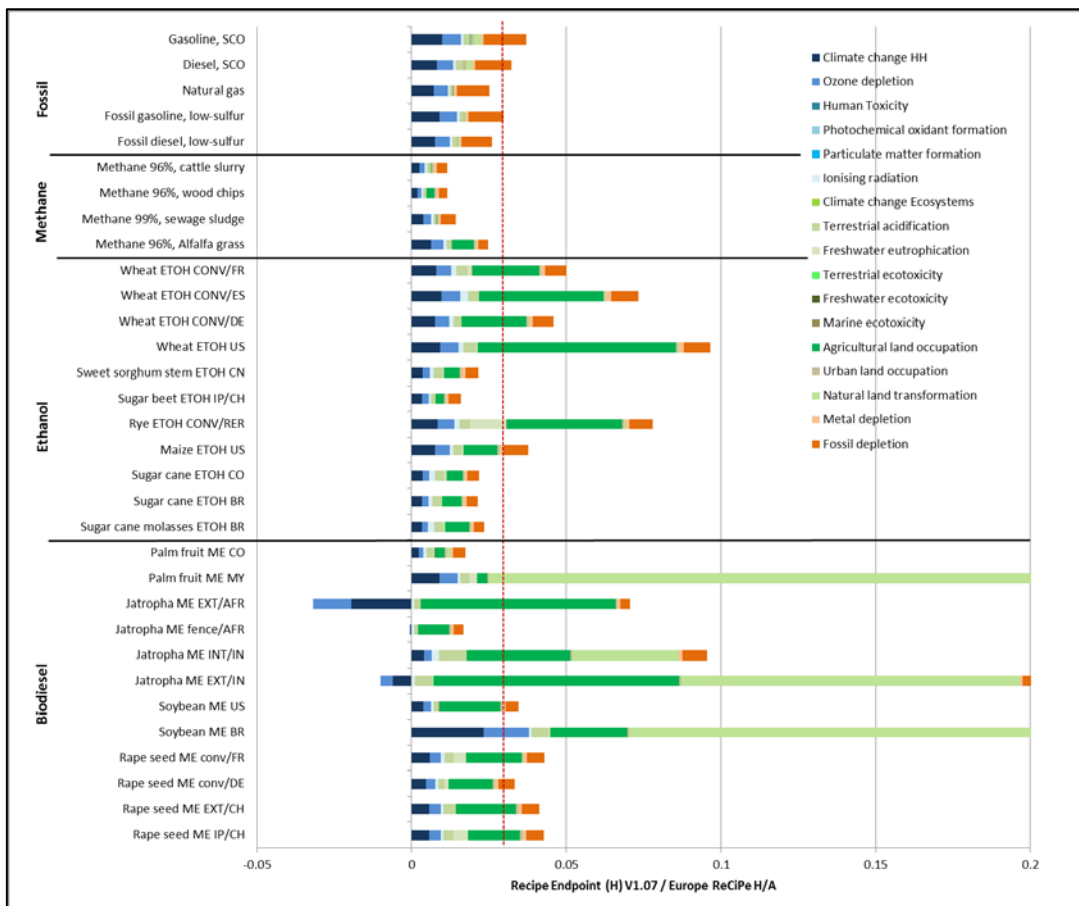
Figure 31:   ReCiPe Europe Endpoint results (points per v.km), by impact category.

4.4. Comparison with 2007study

For this comparison, only pathways assessed in the 2007 study were considered.

4.4.1. GWP results

IPCC 2001
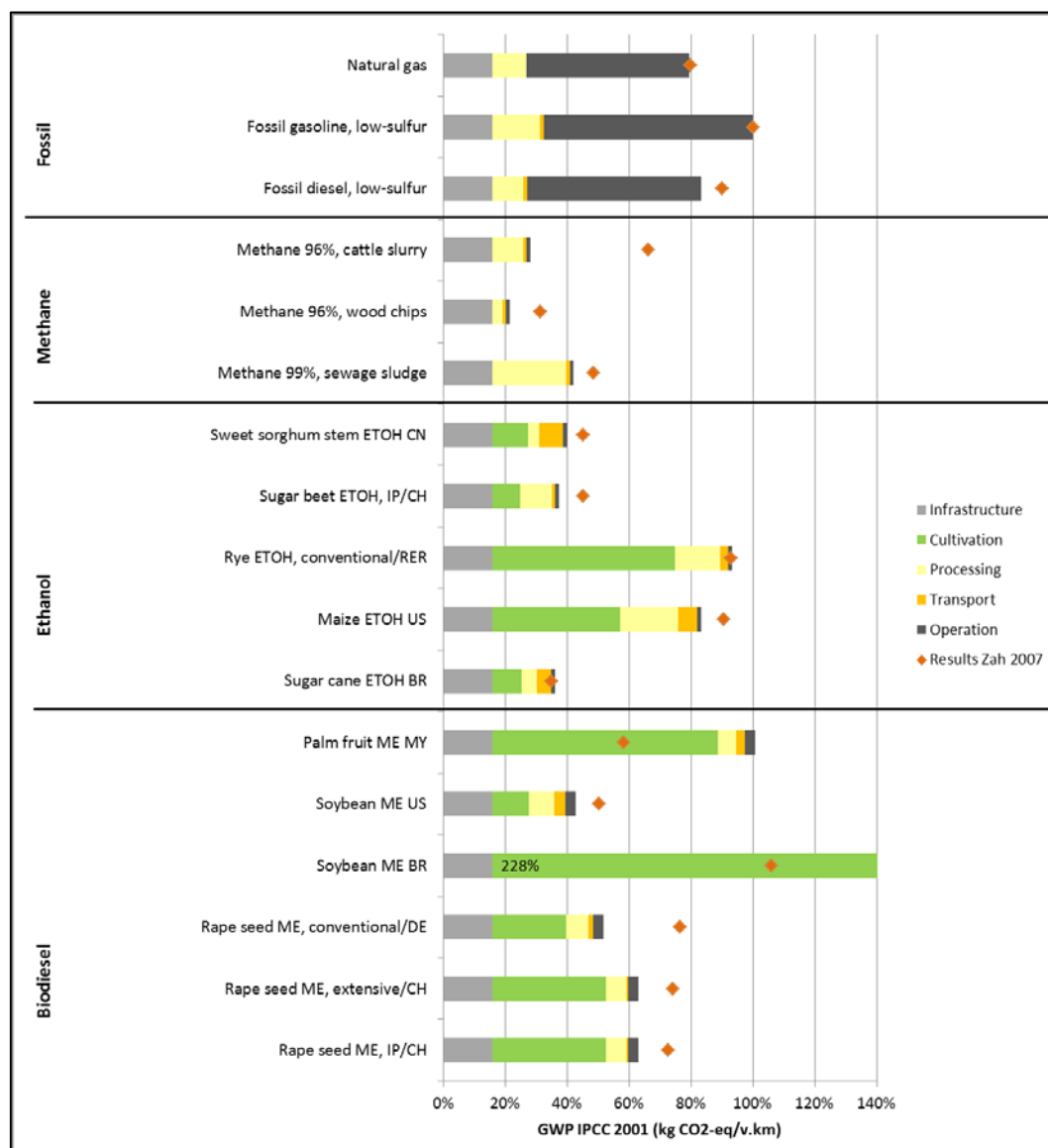
Figure 32: **Relative comparison of the GWP IPCC 2001 results for passenger transport in mid-sized cars with EURO3 emission standard (in kg CO$_2$-eq/v.km) of the 2007 study and of the updated inventories of the same pathways as in 2007. The fossil gasoline represents 100% of the 2007 and the updated results. The result for biodiesel from soybeans BR is overall 228% higher than the reference.**

Figure 32 shows that for most biofuel pathways the relative impacts decreased. The new modelling of nitrogen emissions leads in general to lower results for dinitrogen monoxide, as the emission factors in the IPCC formula are lower (see chapter "Emissions of N$_2$O to the air"). In some cases however, due to the new harmonized calculation, the emissions of nitrate and ammonia, which contribute to the N$_2$O emissions, are higher. This compensates the lower factors for N$_2$O (sugar cane/sugar beets).

In the three pathways sugarcane ethanol (BR, palm fruit (MY) and soybean (BR) methyl ester, the land use change (LUC) emissions were calculated according to the new methodology (see chapter "LUC emissions"). For soybean and palm fruit, the LUC emissions are much higher than in the previous versions. The new accounting of soil carbon and

slashed wood accounts for a part of the increase. In Malaysia, the fact that palm fruit is partly grown on drained peat soil also increases significantly the GHG emissions. Furthermore, new figures concerning the expansion of soybean and palm oil on forest land contribute also to higher GHG emissions. The new discounting of the GHG emissions on 20 years (earlier: 2 years for soybean, 25 years for palm fruit, 1 year for sugarcane) does not compensate this trend.

The updated vehicle operation datasets have higher fuel consumption than in 2007 for the same EURO standard. The increase of fuel consumption is 10% for methane and gasoline, but only 5% for diesel. This accounts for the decrease in the updated results for fossil diesel GWP when compared to gasoline.

The reduction of the emissions in the methane production is due to the new hydrothermal gasification technology which is used in the wood and slurry methane pathways. This pathway does not include any digestion step with the corresponding methane losses.

GHG emissions of the BCM methane from sewage sludge are reduced by about 25% due to the reduction of methane losses in the purification (see also (Lehmann and Zah 2011)).

A direct comparison with the former rape methyl ester pathway "German rape seed methyl ester" is not possible, since the European (RER) rape methyl ester was based on a dataset (rape seed, conventional, Germany) which was replaced in ecoinvent v3 by the former rape seed dataset "rape seed, conventional, Saxony-Anhalt DE". The favourable climatic conditions in Germany lead to low nitrate as well as low dinitrogen monoxide emissions in the German rape seed as compared to Swiss rape seed.

4.4.2. Endpoints with comparison of 2007 results
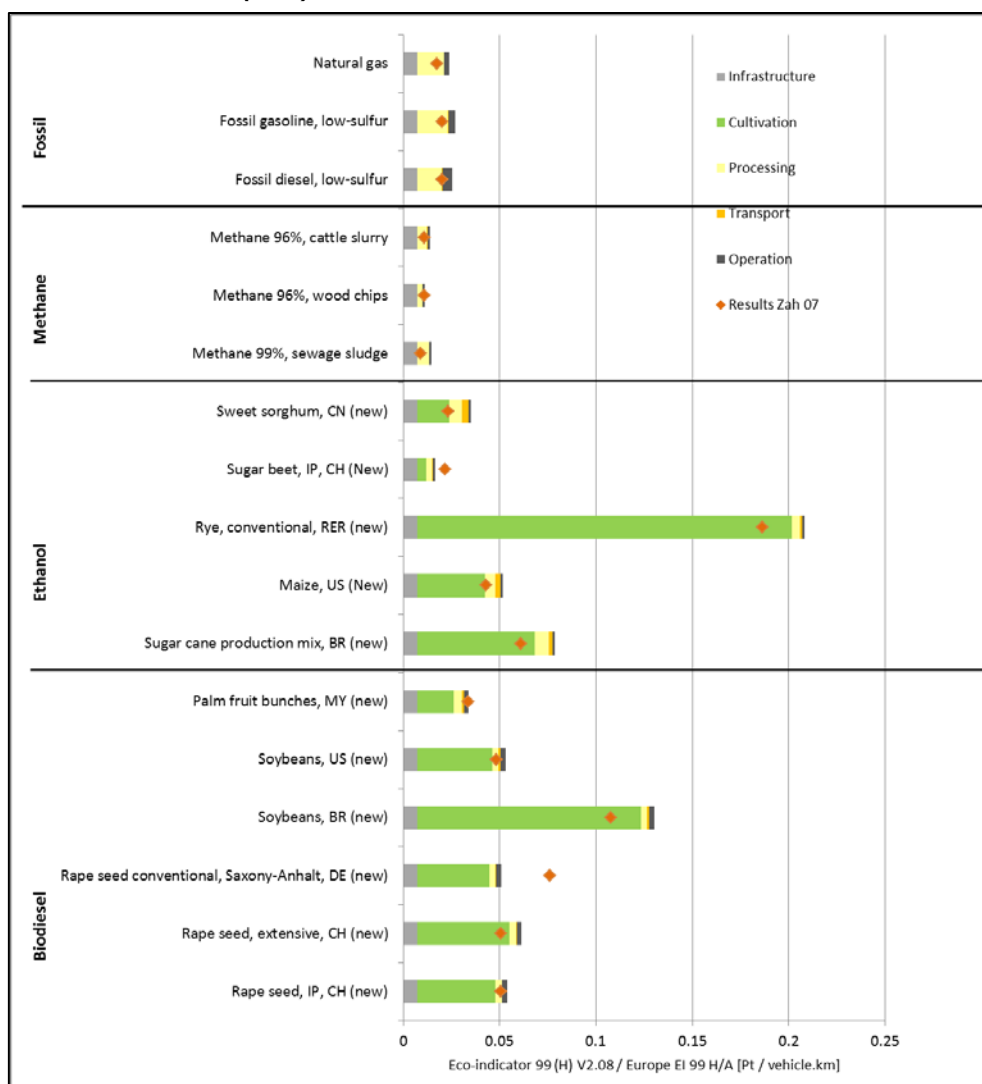
Swiss EcoScarcity 2006 (UBP)
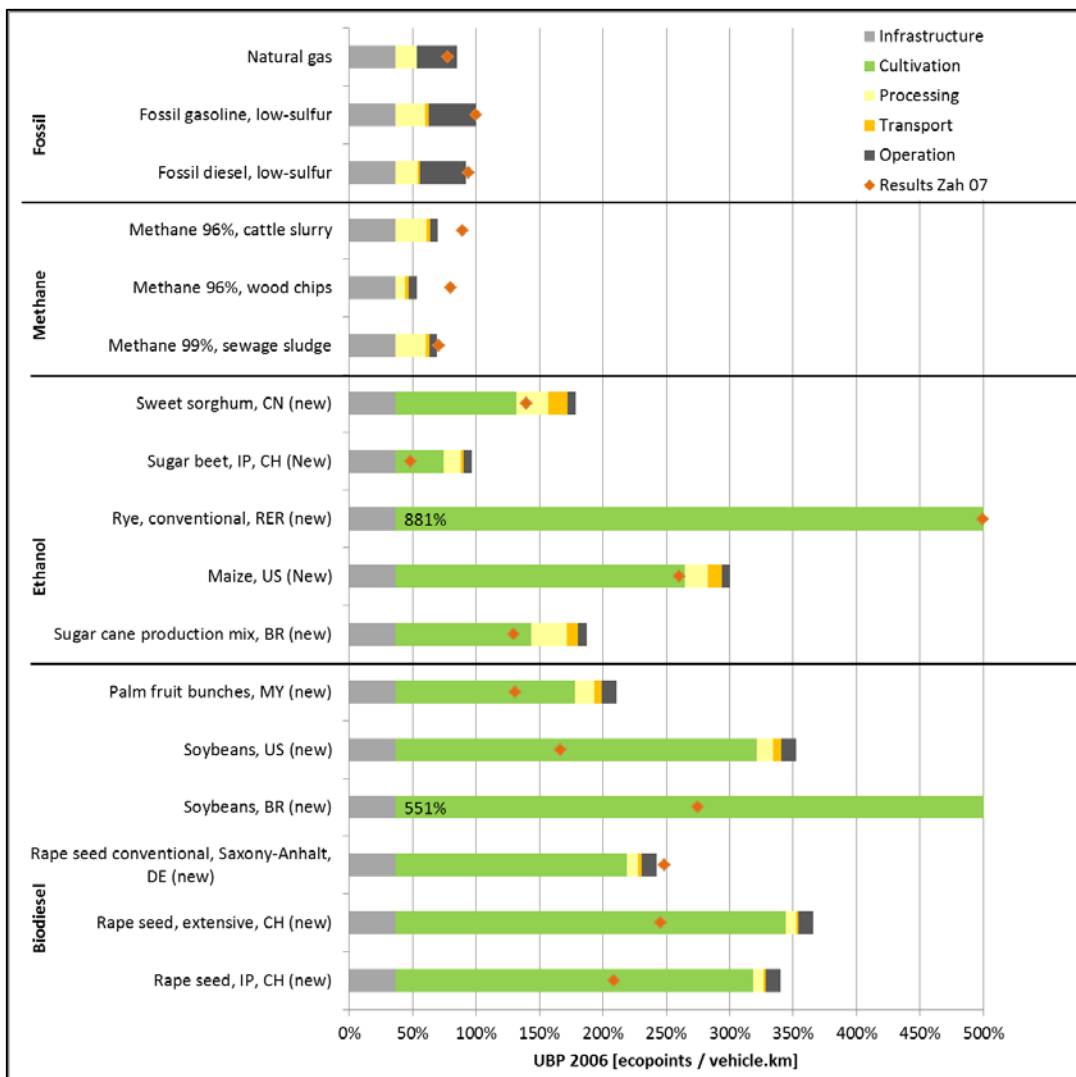
Figure 33: **Relative comparison of the results with the Swiss ecological scarcity method UBP 2006 (in ecopoints/v.km) of the 2007 study and of the updated inventories. The fossil gasoline represents 100% of the 2007 and the updated results.**

The update of the datasets caused an increase of ecological scarcity points for the ethanol and biodiesel pathways and a reduction for the methane pathways. Following parameters influence the results:

- The updated nitrogen emissions lead in most of the cases to higher nitrate emissions and therefore higher UBP values.

- The correction of the pesticide emissions, which were too low in the soybean US dataset, as well as the new impact factor for difluobenzuron emissions (also relevant for soybean ME) leads also to higher results.

- The accounting of heavy metal input on the fields of ashes from biomass combustion and the correction of the heavy metal emissions of sugar beet cultivation, where the uptake from the plant was overestimated, causes higher results for the corresponding pathways.

EcoIndicator '99 (H,A)

Figure 34: Comparison of the results of the 2007 study with the updated inventories with the EcoIndicator '99 (H,A) method (in points/v.km). The fossil gasoline represents 100% of the 2007 and the updated results.

The results with the Ecoindicator method show a similar pattern to those with the Swiss ecological scarcity method. However, the trends are not as pronounced, since nitrate e.g. does not influence as much the results as with the ecological scarcity method.

4.5. Influence of the update of the impact assessment methodology

4.5.1. Climate change indicator

The main differences in the IPCC GWP factor between 2001 and 2007 are the increase of GWP in dinitrogen oxide from 296 to 298 and an increase in the characterisation factor for methane from 23 to 25 CO_2-eq; further some new substances were identified. In our calculations, the overall GWP results increase slightly with the new method where N_2O is important.
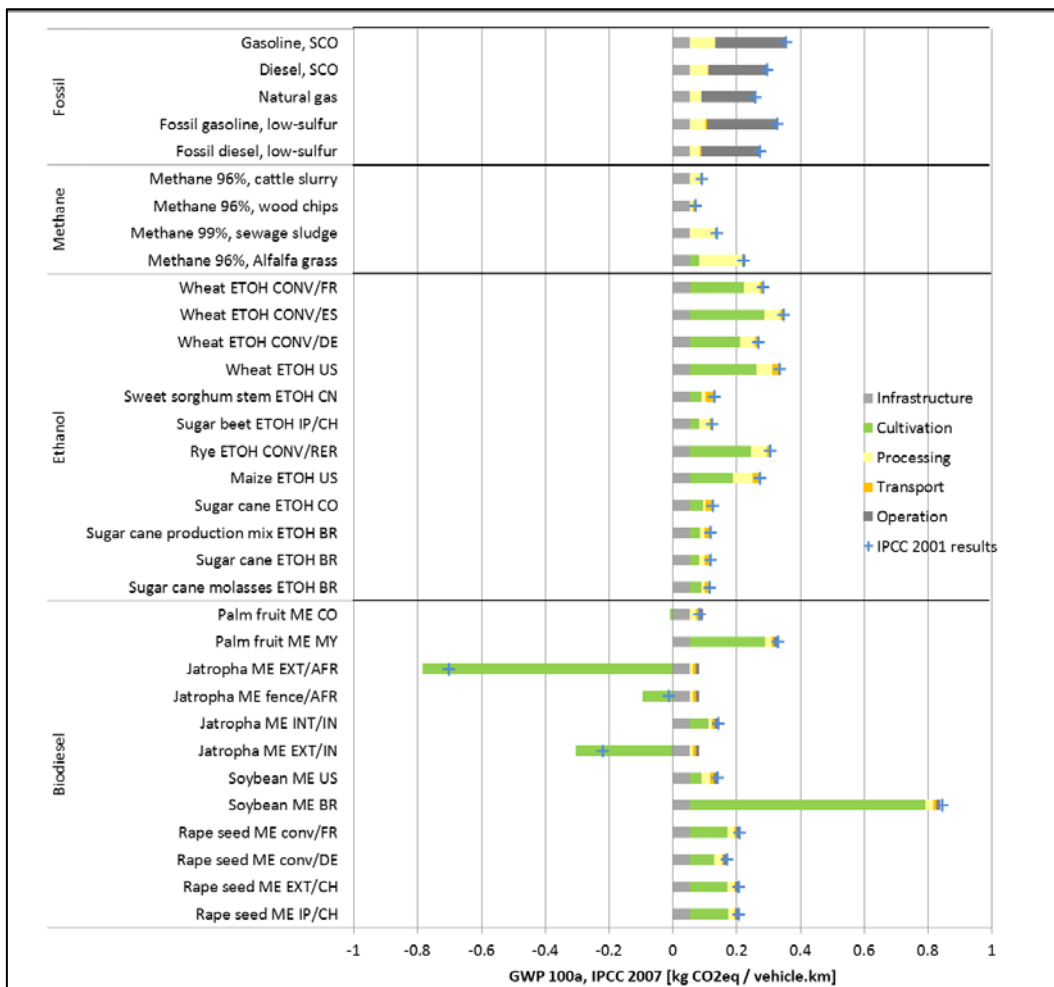
Figure 35: Comparison of the updated inventories calculated with IPCC 2007 (coloured bars) and IPCC 2001 (blue crosses).

4.5.2. Midpoint indicators (CML and ReCiPe)

Here we compare the results of the two different Midpoint assessment methods, CML 2 baseline 2000 (rev.) V2.05 / World, 1990 and ReCiPe Midpoint (H) V1.06 / World ReCiPe H for two selected biodiesel and two selected ethanol production pathways.

The greatest differences between the midpoint results occur in the toxicity indicators (marine aquatic ecotoxicity, fresh water ecotoxicity, terrestrial ecotoxicity, human toxicity). The ReCiPe method uses the new toxicity model, the USES-LCA 2.0 toxicity model, whereas CML relies on an older version of this model.

Eutrophication in ReCiPe is divided in freshwater and marine eutrophication, whereas in CML there is only one cumulated indicator, which explains the lower results in ReCiPe.
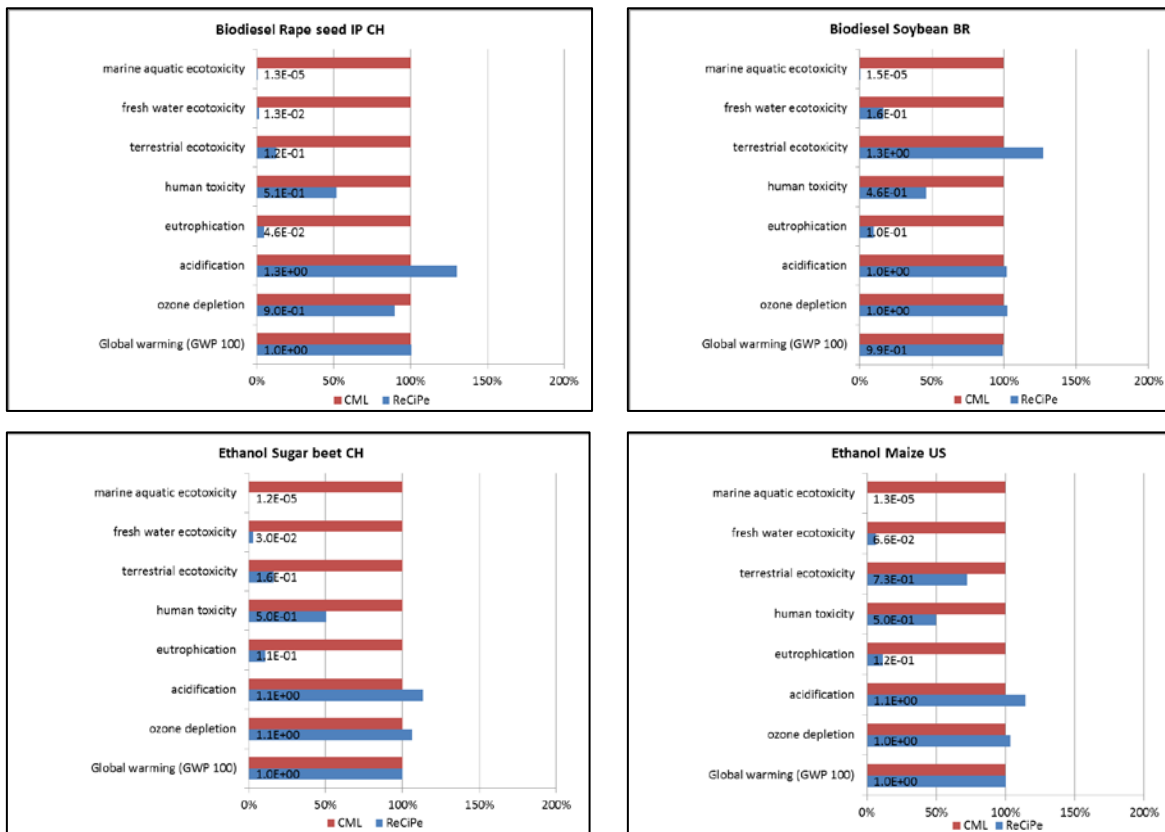
Figure 36: Comparison of the results of the two different Midpoint assessment methods, CML 2 baseline 2000 (rev.) V2.05 / World, 1990 and ReCiPe Midpoint (H) V1.06 / World ReCiPe H for two selected biodiesel and two selected ethanol production pathways.

## 4.6. Uncertainty of the results

The uncertainty sources in LCA are manifold, beginning with the inventory data, which can rely on very different sources, like measurements, modelling or literature data. In many cases the uncertainty of the data cannot be determined from the sample or the source; in the ecoinvent database a pedigree matrix allowing the evaluation of the uncertainty of the flow in a consistent way was developed (see also the quality guidelines of the ecoinvent database) and is used for most of the flows. The LCIA methods have also uncertainties due to the modelling of emission distribution and exposure; however these uncertainties are not systematically documented and therefore currently difficult to take into account.

Here we use a Monte Carlo simulation to calculate the uncertainties. The Monte Carlo analysis (Huijbregts, M. *et al.* 2012)

1. defines uncertainty distributions for input parameters: lognormal, triangular, normal, uniform, etc…

2. translates input uncertainties in output uncertainties by probabilistic sampling: 1,000-10,000 runs

3. visualizes and communicates statistical uncertainty in LCA outcomes.

The following diagrams show the mean values as well as the 95%-confidence interval of the pathways for selected ReCiPe Midpoint indicators and selected pathways. As the uncertainty calculations do not consider the uncertainty of the impact assessment method, the standard deviations reflect the uncertainty of the inventory flows as well as their importance for the overall results. The Monte Carlo analyses included 1'000 runs.

As the distribution of the uncertainty is lognormal, the confidence interval is distributed asymmetrically.
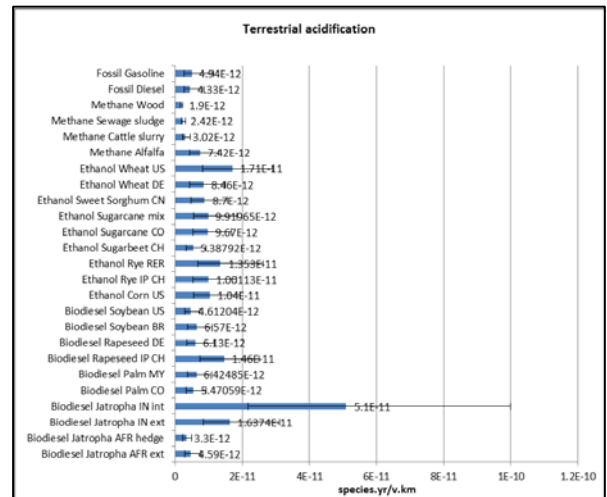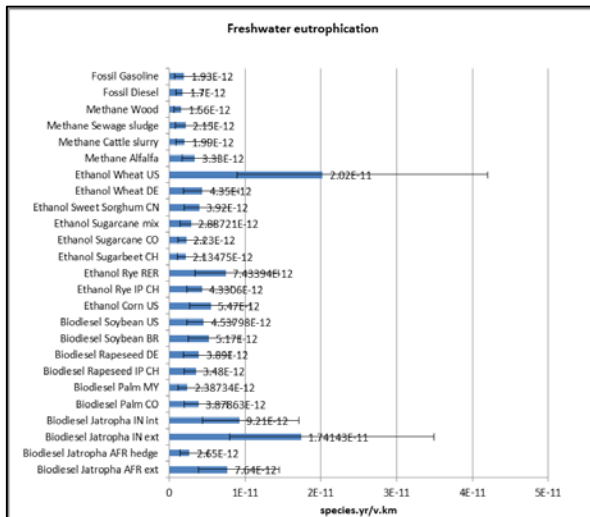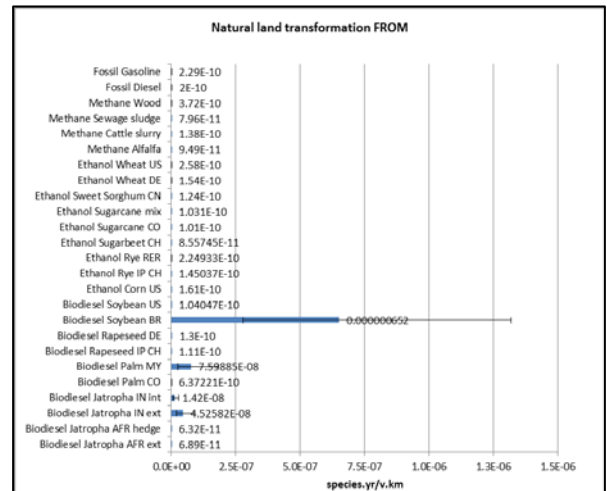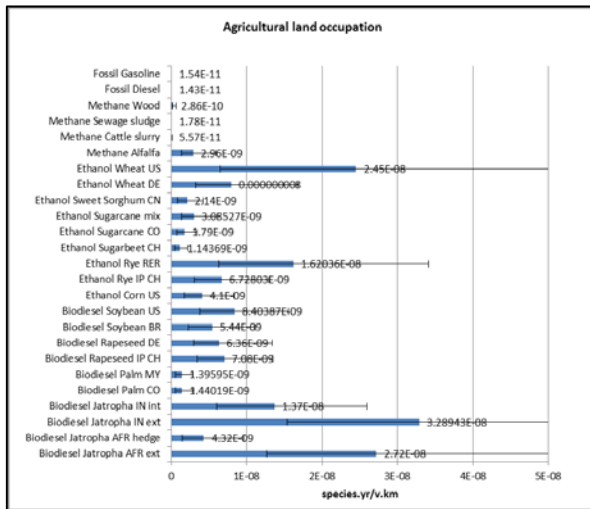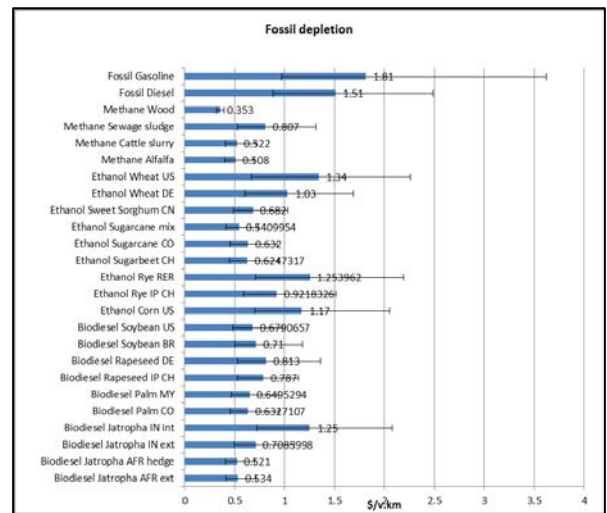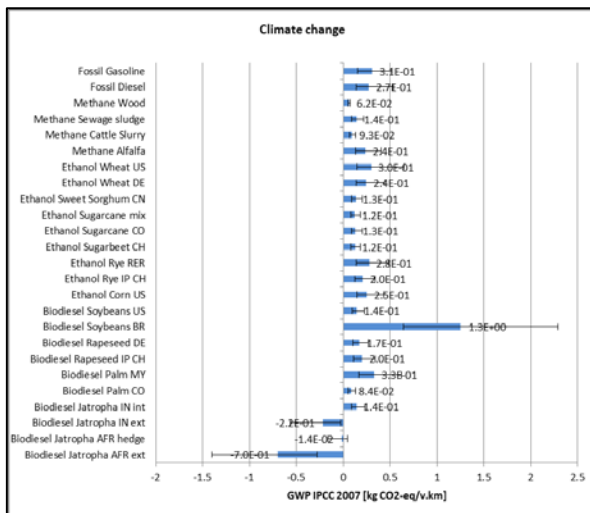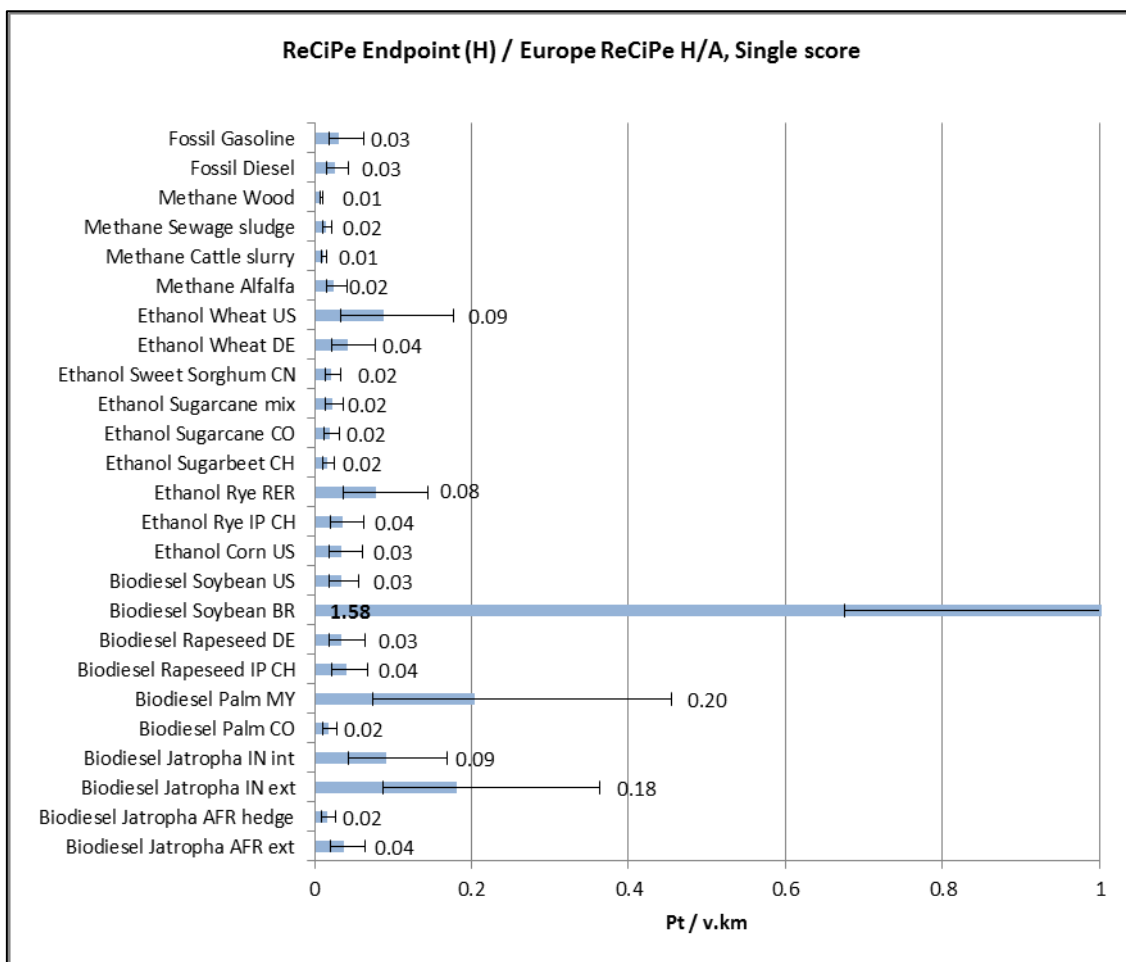
Figure 37: Mean values as well as the 95% confidence interval of the pathways for selected Midpoint indicators
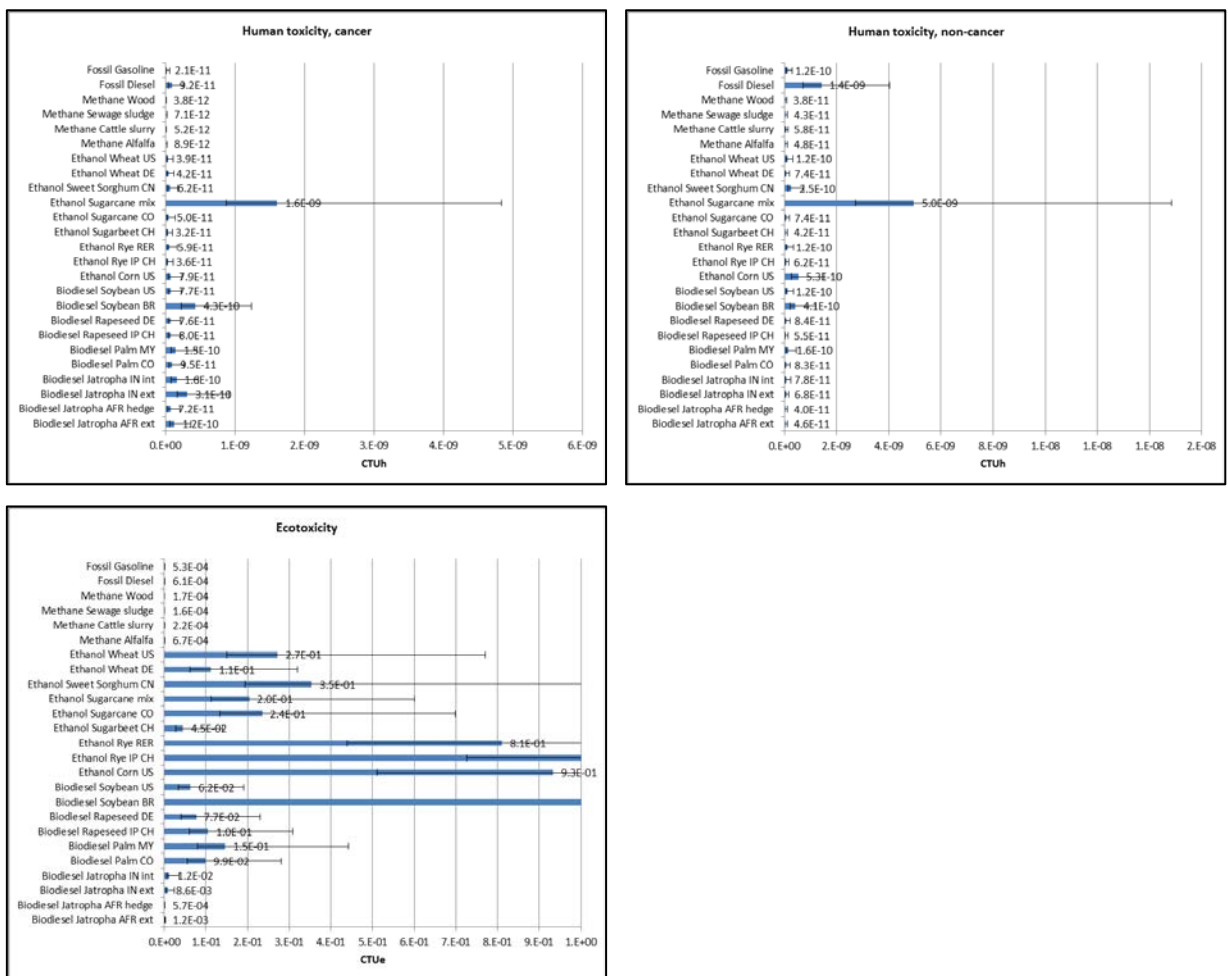
Figure 38: Mean values as well as the 95% confidence interval of the pathways for selected USETox indicators

Figure 39: **Mean values as well as the 95% confidence interval of the pathways for ReCiPe Endpoint Europe H,A results.**

Table 34 shows the coefficient of variation for the selected pathways and the selected indicators. This coefficient varies between 10% and 40% for the single score results. It is usually highest for the agricultural land occupation impacts and lowest for fossil depletion.

Table 34: Coefficient of variation for the selected pathways and selected indicators.

	Single score	Climate change IPCC 2007	Fossil deple-tion	Agricultural land occupa-tion	Natural land transformation FROM	Freshwater eutrophication	Terrestrial acidi-fication
Biodiesel Jatropha AFR ext	30.8%	-43.6%	14.7%	36.3%	23.7%	36.5%	24.6%
Biodiesel Jatropha AFR hedge	31.7%	-378%	14.8%	49%	24.1%	37.3%	21.3%
Biodiesel Jatropha IN ext	39.1%	-64.1%	21.9%	40.1%	40.8%	41.7%	37.4%
Biodiesel Jatropha IN int	35.9%	27.1%	28.5%	39%	39.3%	37%	38.7%
Biodiesel Palm CO	24. 5%	24.3%	19.3%	43.9%	48.9%	36.1%	27.9%
Biodiesel Palm MY	49.2%	35.1%	18.5%	43.0%	53.2%	45.5%	28.2%
Biodiesel Rapeseed IP CH	29%	28.3%	20.3%	35.3%	26.9%	30.4%	31.6%
Biodiesel Rapeseed DE	29.5%	25.8%	21.7%	36.7%	30.7%	45.9%	27.1%
Biodiesel Soybean BR	41.2%	33.5%	23.1%	42.7%	41.4%	37.1%	29.4%
Biodiesel Soybean US	29.8%	24.2%	19.7%	37.4%	30.3%	33.8%	26.3%
Ethanol Corn US	31.5%	29.9%	29.1%	42.3%	38%	37.8%	33.5%
Ethanol Rye IP CH	29.7%	27.3%	25.1%	36.4%	34.5%	34.5%	31.3%
Ethanol Rye RER	36.8%	33%	29.6%	44.3%	39.2%	37.6%	35.3%
Ethanol Sugarbeet CH	22.6%	21.4%	19.6%	38.1%	29.4%	63.3%	28.7%
Ethanol Sugarcane CO	25.9%	23.6%	19%	40.6%	31.4%	42.9%	32.5%
Ethanol Sugarcane mix	26.4%	22.6%	15.5%	39.0%	30.0%	40.2%	32.2%
Ethanol Sweet Sorghum CN	26%	23.4%	20.3%	42.6%	34.5%	38.3%	34.6%
Ethanol Wheat DE	32.1%	30.5%	26.4%	38.8%	45.1%	40.1%	32.5%
Ethanol Wheat US	39.5%	36%	28.7%	50.3%	37.9%	37.4%	32.3%
Methane Alfalfa	28.2%	30.2%	14.8%	39.5%	30.8%	38.7%	30.9%
Methane Cattle slurry	15.6%	16.7%	14.4%	61.2%	47.8%	45.9%	18.6%
Methane Sewage sludge	22.4%	22.1%	25.6%	41.2%	27.3%	51.5%	14%
Methane Wood	10.9%	8.33%	5.65%	47.7%	49.3%	46.5%	8.85%
Fossil Diesel	26%	34.3%	26.6%	34.7%	64.5%	47.3%	38%
Fossil Gasoline	34.3%	28.3%	35.4%	37.3%	71.1%	57.1%	52.2%

4.7. Discussion

4.7.1. Changes from Zah et al. 2007 to the inventories 2012

This study uses harmonized and corrected inventories, therefore allowing coherent and well-grounded comparison of biofuel pathways. The harmonization of the modelling of N-emissions mostly leads to lower impacts for the biofuel pathways, mainly due to lower N_2O-emissions. However, the results of the new modelling of nitrate and ammonia do not show any trend so that in some cases they compensate the lower factor for N_2O emissions in the IPCC formula.

The update of the vehicle emissions and fuel consumption on the basis of more recent models and publications (see chapter "Conversion processes") leads to higher impacts for the car operation and also higher impacts for the pathways due to higher consumption; both factors partly overcompensate the reduction described above.

4.7.2. Results of new pathways (jatropha/methane)

The assessment of jatropha shows distinctly that even for this feedstock, which is sometimes described as "wonder crop", the overall results depend strongly on the cultivation system and practice. The best results are achieved by jatropha planted as a fence, because of the low inputs in cultivation. However, this option can only be realised for a local market, as the yields are very low. Furthermore, those results are only valid if jatropha is grown on fallow land. As soon as jatropha is cultivated intensively, the results of the pathway are very similar to other feedstocks.

The new methane pathways show low impacts when compared to fossil fuels and other biofuels. This is due partly to the fact that two of the pathways are based on waste materials (sewage sludge, slurry) which carry no impacts when entering the system. The pathways based on wood and alfalfa have higher impacts caused by the cultivation phase resp. forestry activities. However, the overall results are still better than those of most other biofuel pathways. These results confirm the observations made in the 2007 study. The optimization of the pathways with regard to methane losses compensates the higher vehicle consumption of the updated car operation inventories.

4.7.3. Changes in the fossil reference

The evaluation of the production of fossil fuels from unconventional oil, here with the example of oil sands, is only partial because the tailings could not be included due to lack of resources and data; the tailings are expected to increase the toxicity indicators.

Most of the GHG emissions of fossil fuel use occur during combustion. Therefore, our provisory results show that the higher impacts on global warming caused during the extraction and processing of these fuels are buffered by the emissions of combustion when compared on a vehicle-kilometre basis. For natural land transformation and water depletion, however, the indicator results are ten to hundred times higher in the case of fuel from oil sands compared to conventional fuels. This substantiates the current environmental concerns about these fuels.

4.7.4. Land use change emissions

This study presents a comprehensive and coherent assessment of land use change emissions. All compartments of carbon stocks (from above ground biomass to soil carbon) are considered and assessed with a methodology in accordance to IPCC 2007. The quantification of the land use change areas and the causality are assessed based on the recommendations of (Milà i Canals, L. *et al.* 2012) and are based on averaged data over 20 years.

With this methodology, the emissions from land use change account to a great extent for the environmental impacts of the biofuel pathways. In fact, if biofuel crops are grown on a previous rain forest area, the pay-back period for the provoked GHG emissions due to the land transformation can be very long. The difficulty of this assessment lies in establishing the causality between the deforestation and the cultivation of a certain crop. Furthermore, feedstocks for biofuel can be grown on pasture which had been deforested some years before; or they can grow on former cropland or pasture land and thus induce expansion of

the displaced fodder production on forest area. This effect is called indirect land use; at this time, there is no agreed method to account for indirect land use emissions.

4.7.5. Emissions caused by indirect land use change (iLUC)

There is a trade-off between assessing LUC and iLUC. If the direct LUC-GHG emissions have to be included in the LCA of biofuels as it is e.g. the case for the RED methodology, new energy plantations will be created on agricultural land, because the direct LUC factors can be omitted.

Figure 40 illustrates the potential iLUCs from oil palm cultivation with an example from Colombia. The calculations account for two cases where palm is cultivated either on former grassland or on agricultural land. The displacement leads to a pressure on natural land (rainforest, moist deciduous forest or scrubland). Depending on what natural system is affected, the iLUC has a significant impact on the carbon stock and hence on the GWP.

If tropical rainforest or moist deciduous forests are displaced indirectly, the GWP of biofuels is much higher compared to fossil fuels. If scrubland is displaced, the impacts are lower and the GHG balance of the biofuels is still positive compared to that of fossil fuels (Gmünder, S. *et al.* 2012).
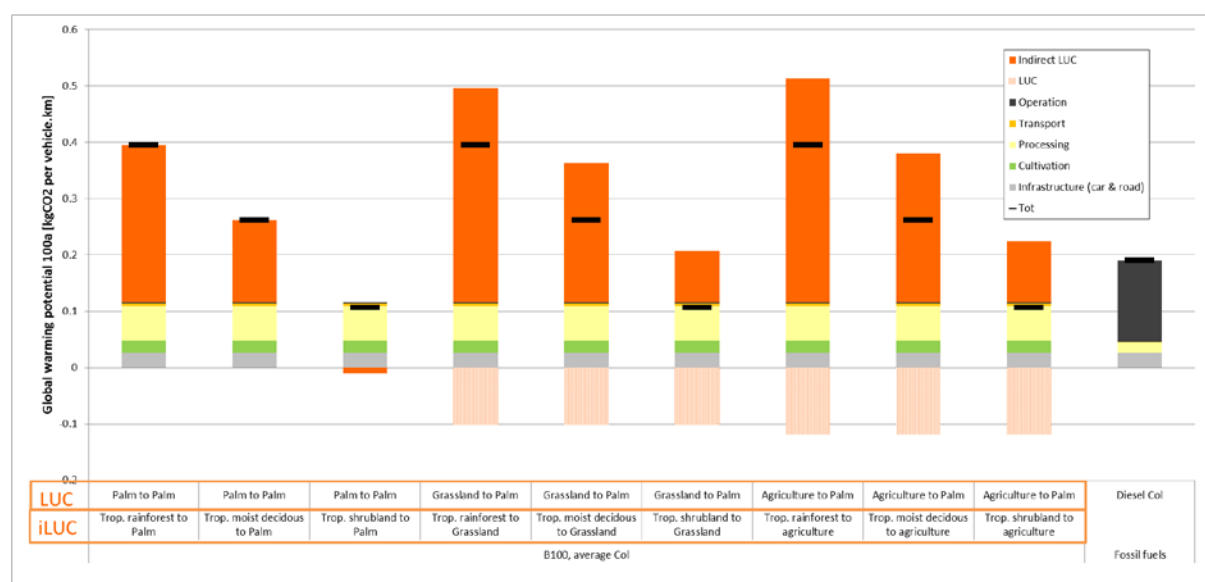
Figure 40: Potential indirect land use impacts of oil palm cultivation in Colombia (Gmünder, S. *et al.* 2012).

While direct land use change can be directly linked to a biofuel project, iLUC has to be modelled and is more difficult to allocate to a specific crop. Moreover, iLUC can happen in another country because the change in demand can influence the global market.

A possible way of dealing with this problem can be to account iLUC with an iLUC-GHG-risk factor. However, this approach can penalize projects which implement measures to prevent iLUC. iLUC can indeed be reduced if specific measures are implemented (expansion on underutilised land).

Taking into account only direct land use change favours European and US biofuel production, because no direct land use change occurs in Europe or in the US. However, the increased production can induce a higher demand for other agricultural products on the global market and therefore cause indirect land use effects.

4.7.6. Influence of assessment method

All methods agree on the fact that the cultivation phase is one of the most important phases in the production of biofuels. For waste based biofuels, the processing phase dominates. While the ranking of the biofuels stays mostly stable with the different endpoint assessment methods, the comparison with fossil fuels yields different results.

The main motivation for producing biofuels is the replacement of fossil fuel and consequently reducing fossil CO_2-emissions by relying on a biomass-based system creating mainly

biogenic CO_2-emissions. However, because of the N_2O-emissions caused by agriculture and the high reliance of cultivation methods on mineral fertilizer and machinery using fossil fuel, the greenhouse gas emissions of the agricultural phase can be high, depending on the pathway, and therefore even not allow any reduction when compared to fossil fuels.

In this study we use as in 2007 the simplified method where biogenic CO_2 emitted by the biofuel combustion is equal to the assimilated CO_2 and can therefore be considered as neutral. However, the EEA criticizes this assumption as having the wrong baseline. For the EEA, only assimilated carbon which would not have been assimilated without the biofuel project can be considered as CO_2-neutral.

Depending on the assessment methodology, the results are rather in favour of the fossil fuels or of the biofuels. Ecoindicator 99 is very sensitive to land use, whereas the ecological scarcity method reacts strongly to changes in nitrate or heavy metal emissions. In the ReCiPe method, the depletion of fossil resources is weighted very strongly in consequence of the modelling chosen for resource depletion. The evaluation of the Midpoint indicators shows, however, that the only indicators where biofuel pathways based on agricultural crops always perform better are fossil depletion and ozone depletion. In all other categories, the values for biofuel pathways can be significantly higher than the fossil reference. These results indicate a problem shifting when promoting biofuels for their reduced GHG emissions, which is also the outcome of a recent paper on corn ethanol (Yang, Y. *et al.* 2012).

4.7.7. Allocation: choice of method, system boundaries

The modelling of co-producing activities can be done with different methods and around different system boundaries, which yield different results. However, the energetic allocation approach, while based on more stable parameters, also presents relevant modelling uncertainties and does not reflect the drivers of the processes. The results of this study are provisory and not shown here. Indeed the different allocation methods were only used in foreground processes and not on all processes involved. Such a modelling will only be possible with ecoinvent v3, which will be released in September 2012.

4.8. Conclusions

Although biofuels can lead to a reduction of fossil fuel use and of greenhouse gas emissions, they often shift environmental burdens towards land use-related impacts; indeed, only very few biofuel pathways show lower or at least no higher impacts than the fossil fuels for all indicators. The most promising pathways are those based on methanisation of residues or on reforestation activities.

The study confirms the high diversity in the impact patterns of biofuel pathways and therefore the necessity of assessing biofuel projects with specific data. The uncertainty of the results is high due to lack of data and modelling uncertainties. There is for example a need for more specific modelling of agricultural N_2O. This uncertainty should lead to general caution when promoting biofuels.

References

Achten, W. M. J., L. Verchot, Y. J. Franken, E. Mathijs, V. P. Singh, R. Aerts and B. Muys (2008). "Jatropha bio-diesel production and use." Biomass and Bioenergy **32**(12): 1063-1084.

Agrammon group (2009a). Technical process description AGRAMMON – Draft. Online at: www.agrammon.ch.

Agrammon Group (2009b). Technische Parameter Modell Agrammon. Schweizerische Hochschule für Landwirtschaft SHL. Online at: www.agrammon.ch.

Allen, E. W. (2008). "Process water treatment in Canada's oil sands industry: I. Target pollutants and treatment objectives." J. Environ. Eng. Sci. **7**: 123-138.

Althaus, H.-J., C. Bauer, G. Doka, R. Frischknecht, N. Jungbluth, T. Nemecek, A. Simons, M. Stucki, J. Sutter and M. Tuchschmid (2010). Documentation of changes implemented in ecoinvent Data v2.1 and v2.2. Data v2.2 (2010) ecoinvent report No. 16. . ecoinvent report R. H. (Editor). St. Gallen, ecoinvent Centre

Angele, H.-C. (2006). Was tun mit den biogenen Abfällen in der Stadt Zürich? Kommentare zur Studie von Entsorgung und Recycling Zürich (ERZ) – Argumentarium für eine Vergärung. http://www.biomasseenergie.ch/dt/pdf/argumentarium%20Zueridef060406.pdf (10 Apr 2006). Zollikon, CH, Bundesamt Für Energie, Programm Energie Schweiz, BiomassEnergie.

BAFU (2008). Abfallwirtschaftsbericht 2008 – Zahlen und Entwicklungen der schweizerischen Abfallwirtschaft 2005–2007. Umwelt-Zustand Bericht UZ 0830. Bundesamt für Umwelt (BAFU) Autoren: Hügi M., Gerber P., Hauser A., Laube A., Quartier R., Schenk K, Wysser M. http://www.bafu.admin.ch/publikationen/publikation/01009/index.html?lang=de&downloadshop=NHzLpZig7t,lnp6I0NTU042l2Z6ln1acy4Zn4Z2qZpnO2Yuq2Z6gpJCDdIN4fGym162dpYbUzd,Gpd6emK2Oz9aGodetmqaN19XI2IdvoaCVZ,s-.pdf (26 Nov 2009)

Bern, Schweiz, BAFU.

BAFU (2010). Zeitreihe brennbare Abfälle und KVA-Kapazität, 2001 bis 2009", June 2010. http://www.bafu.admin.ch/abfall/01517/01519/10457/index.html?lang=de&download=NHzLpZeg7t,lnp6I0NTU042l2Z6ln1acy4Zn4Z2qZpnO2Yuq2Z6gpJCGd3x,fWym162epYbg2c_JjKbNoKSn6A (6 July 2010). Bern, Schweiz, Bundesamt für Umwelt (BAFU)

BAFU (2011). Abfallstatistiken. http://www.bafu.admin.ch/abfall/01517/01519/index.html?lang=de (17 Feb 2011). Bern, Schweiz, Bundesamt für Umwelt (BAFU)

Banks, C. (2004). Biodigestion of kitchen waste – A comparative evaluation of mesophilic and thermophilic biodigestion for the stabilisation and sanitisation of kitchen waste. Final report, The University of Southampton and Greenfinch Ltd. http://www.soton.ac.uk/~sunrise/Greenfinch.htm and http://www.soton.ac.uk/~sunrise/Biodigestion%20final%20report.pdf (9. Nov 2010).

BFE (2010). Schweizerische Gesamtenergiestatistik 2009. July 2010 http://www.bfe.admin.ch/php/modules/publikationen/stream.php?extlang=de&name=de_76983770.pdf&endung=Schweizerische%20Gesamtenergiestatistik%202009 (24 Sept 2010). Berne, Switzerland, Bundesamt für Energie BFE.

Bilitewski, B., B. Härdtle and K. Marek (1991). Abfallwirtschaft – Eine Einführung. Berlin Heidelberg New York, Springer Verlag.

Biomasse Schweiz (2006). Strom- und Wärmeproduktion aus Grüngut: Vergärung oder Mitverbrennen in KVA. Vorabzug aus der PSI-Studie im Auftrag von Biomasse Schweiz. 5. Aprill 2006 http://www.biomasseenergie.ch/dt/pdf/Vorabzug%20KVA%20vs%20Verg%E4rung%20_4_%20Welllinger.pdf (10. April 2006). Aadorf, Switzerland, BiomasseSchweiz.

Brandão, M. and L. i Canals (2012). "Global characterisation factors to assess land use impacts on biotic production." The International Journal of Life Cycle Assessment: 1-10.

Brisson, N., C. Gary, E. Justes, R. Roche, B. Mary, D. Ripoche, D. Zimmer, J. Sierra, P. Bertuzzi, P. Burger, F. Bussière, Y. M. Cabidoche, P. Cellier, P. Debaeke, J. P. Gaudillère, C. Hénault, F. Maraux, B. Seguin and H. Sinoquet (2003). "An overview of the crop model STICS. ." European Journal of Agronomy **18**: 309–332.

Brown, G. (2008). Review of fuel ethanol impacts on local air quality. Bioethanol for sustainable transport (BEST) Deliverable No D9.14. London, Imperial Centre for Energy Policy and Technology (ICEPT).

BSI (2011). The Guide to PAS 2050:2011. How to carbon footprint your products, identify hotspots and reduce emissions in your supply chain. London, British Standards Institution

BUWAL (1995). Zusammensetzung der Siedlungsabfälle der Schweiz 1992/93. Berne, Switzerland, Schriftenreihe Umwelt Nr. 248.

BUWAL (1997). Dioxine und Furane – Standortbestimmung, Beurteilungsgrundlagen, Massnahmen. Autoren: Dettwiler J., Karlaganis G., Studer C., Joss S., Stettler A., Chambaz D. . http://www.bafu.admin.ch/publikationen/publikation/00429/index.html?lang=en. Berne, Schweiz, Schriftenreihe Umwelt Nr. 290, Bundesamt für Umwelt, Wald und Landschaft BUWAL (now BAFU)

Carmenza, R. and J. Blaser (2008) "Key Issues on Land Use, Land Use Change and Forestry (LULUCF) with an Emphasis on Developing Country Perspectives. An Environment & Energy Group Publication." **Volume**, 49 DOI:

Caslin, B., J. Finnan and L. Easson (2011). MISCANTHUS BEST PRACTICE GUIDELINES. Carlow & Hillsborough, Crops Research Centre & Agri-Food and Bioscience Institute.

CML (2001). CML-IA Characterisation Factors. http://cml.leiden.edu/software/data-cmlia.html. Leiden, Institute of Environmental Sciences (CML).

Dahl, J. and I. Obernberger (2004). EVALUATION OF THE COMBUSTION CHARACTERISTICS OF FOUR PERENNIAL ENERGY CROPS (ARUNDO DONAX, CYNARA CARDUNCULUS, MISCANTHUS X GIGANTEUS AND PANICUM VIRGATUM). 2nd World Conference on Biomass for Energy, Industry and Climate Protection, Rome.

Dawson, M. (2011). SHORT ROTATION COPPICE WILLOW BEST PRACTICE GUIDELINES. B. CASLIN, FINNAN, J., McCRACKEN, A. . Belfast, Agriculture and Food development Authority.

De Willigen, P. (2000). An analysis of the calculation of leaching and denitrification losses as practised in the NUTMON approach. Report 18. Wageningen, The Netherlands, Plant Research International.

Dean, A., S. Hughes, W. Gibbons, A. Syed, O. Tsui, C. Renou, D. Dow, A. Mangin and T. Boivin (2007). MONITORING SUSTAINABLE DEVELOPMENT IN THE OIL SANDS REGION OF ALBERTA, CANADA, USING SPOT-5 AND ENVISAT ASAR AND MERIS IMAGERY. 'Envisat Symposium 2007', 23–27 April. Montreux, Switzerland.

Dettli, R., P. Steiner, M. Baur and M. Müller (2004). Kosten und Entschädigung von Elektrizität aus Kehrichtverbrennungsanlagen. http://www.energie-schweiz.ch/imperia/md/content/politikundrecht/energiepolitik/ewg/51.pdf (4 July 2006). Bern, CH, Bundesamt für Energie BFE.

DGE (2004). Biogas aus Abfall zur Erzeugung von Strom und Erdgas. Lutherstadt Wittenberg, DE, Dr.-Ing. Günther Engineering GmbH.

Doka, G. (2007). Waste Management Services. ecoinvent Report No. 17 part II.v. (pp.628-683) for the ecoinvent database v2.0. Online-Version freely available for registered guests and members at http://www.ecoinvent.org/documentation/reports/. Dübendorf, CH, Swiss Centre for Life Cycle Inventories.

Dones, R., Ed. (2007). <u>Sachbilanzen von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz</u>. ecoinvent report No. 6. Duebendorf, CH, Swiss Centre for Life Cycle Inventories.

EC-JRC-IES (2011). International Reference Life Cycle Data System (ILCD) Handbook-Recommendations for Life Cycle Impact Assessment in the European context. First edition November 2011. <u>ILCD handbook</u>. Luxemburg, European Commission, Joint Research Centre, Institute for Environment and Sustainability.

EC-JRC-IES (2012). Characterisation factors of the ILCD Recommended Life Cycle Impact Assessment methods. Database and Supporting Information. First edition. February 2012. EUR 25167. <u>Publications Office of the European Union</u>. Luxembourg, European Commission, Joint Research Centre, Institute for Environment and Sustainability.

Edelmann, W. (2008). Bestimmung der TS- und OS-Gehalte von Ausgangsmaterialien für die Feststoffvergärung. Arbeitsgemeinschaft Bioenergie arbi, BFE Projekt 102112 / 152631. November 2008. http://www.biogas.ch/images/stories/pdf/ts_ausgangsmaterialien.pdf (9. Aug 2010). Bern, CH, Bundesamt für Energie.

Edelmann, W. and K. Schleiss (2001). Ökologischer, energetischer und ökonomischer Vergleich von Vergärung, Kompostierung und Verbrennung fester biogener Abfallstoffe. Commissioned by Bundesamt für Energie (Swiss Federal Office of Energy) and BUWAL, Bundesamt für Umwelt, Wald und Landschaft (Swiss EPA, SAEFL). http://www.naturemade.org/common/texted_zertifizierung/%C3%96kobilanz%20Feststoffverg%C3%A4rung.pdf (10 Juli 2005) http://www.bfe.admin.ch/php/modules/enet/streamfile.php?file=000000007031.pdf&name=210031.pdf (9. Nov 2010). Bern, CH, Bundesamt für Energie & Bundesamt für Umwelt, Wald und Landschaft.

Eggleston, H. S., L. Buendia, K. Miwa, T. Ngara and K. Tanabe, Eds. (2006). <u>2006 IPCC Guidelines for National Greenhouse Gas Inventories.</u> Hayama, Japan.

Egli, S. (2005). Anleitung mit Checkliste zur Energieoptimierung von Kehrichtverbrennungsanlagen. S&A Engineering AG, Schaffhausen. Commissioned by Swiss Federal Office of Energy (SFOE) (BFE). DIS-Project No. 100686. Final report March 2005. http://www.bfe.admin.ch/php/modules/enet/streamfile.php?file=000000008973.pdf&name=000000250054.pdf (5. July 2006). Bern, CH, Swiss Federal Office of Energy

EMPA (2009). Ökobilanz: Biomethan-Aufbereitungsanlage Meilen. Dübendorf, CH, EMPA, Technology and Society Lab.

Entsorgung + Recycling Zürich (2006). Verwertung der biogenen Abfälle in der Stadt Zürich. Entsorgung + Recycling Zürich ERZ, Stadt Zürich. Switzerland. 6.April 2006, http://www3.stzh.ch/internet/erz/home/khkw/Biogene_Abfaelle.ParagraphContainerList.ParagraphContainer1.ParagraphList.0002.File.pdf/Bericht%20Biogene%20Abf√§lle%20der%20STZH%20Publikation%20Druckversion.pdf () oder http://www.stadt-zuerich.ch/content/dam/stzh/ted/Deutsch/erz/Kehrichtheizkraftwerke/Publikationen_und_Broschueren/KEH_Biogene%20Abfaelle_0604.pdf (6 Apr 2006). Zürich, ERZ.

Environment-Canada (2009a). National Polutant Release Inventory: 2008 Facility & Substance Information for Shell Canada Energy, Shell Albian Sands Muskeg River Mine. Gatineau, Environment-Canada.

Environment-Canada (2009b). National Polutant Release Inventory: 2008 Facility & Substance Information for Shell Canada Limited - Scotford Upgrader. Gatineau, Environment-Canada.

EU-Commission (2010). Commission Decision of [date] on guidelines for the calculation of land carbon stocks for the purpose of Annex V of Directive 2009/28/EC. Brussel, EC.

Fahrni, H. P. (2006). Probleme und Fragestellungen aus Sicht der Praxis in der Abfallwirtschaft - Ansätze zur Lösung. Lecture at ETH Zurich 26th June 2006. http://www.uns.ethz.ch/edu/teach/EinfAnthr/downloads/VL06/VL_07hpf.

Faist Emmenegger, M., T. Heck, N. Jungbluth and M. Tuchschmid (2007). Erdgas. Sachbilanzen von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. Final report ecoinvent No. 6-V. R. E. e. a. Dones. Villigen& Dübendorf, CH, Paul Scherrer Institut&Swiss Centre for Life Cycle Inventories.

Faist Emmenegger, M., J. Reinhard and R. Zah (2009). SQCB - Sustainability Quick Check for Biofuels. Second draft, 18th February 2009 - Intermediate Background Report. With contributions from T. Ziep, R. Weichbrodt, Prof. Dr. V. Wohlgemuth, FHTW Berlin and A. Roches, R. Freiermuth Knuchel, Dr. G. Gaillard, Agroscope Reckenholz-Tänikon. . Dübendorf, Empa.

FAOSTAT (2012). Agriculture Data, Food and Agriculture Organsiation of the United Nations.

Fargione, J., J. Hill, D. Tilman, S. Polasky and P. Hawthorne (2008). "Land Clearing and the Biofuel Carbon Debt." Science **319**(5867): 1235-1238.

Friedl, A., E. Padouvas, H. Rotter and V. K. (2005). "Prediction of heating values of biomass fuel from elemental composition." Analytica Chimica Acta **544**: 191-198.

Frischknecht, R., A. Braunschweig, H. P. and S. P. (2000). "Modelling human health effects of radioactive releases." Life Cycle Impact Assessment. Environmental

Impact Assessment Review **20** (2): 159-189.

Frischknecht, R., R. Steiner and N. Jungbluth (2008). Ökobilanzen: Methode der ökologischen Knappheit - Ökofaktoren 2006. Methode für die Wirkungsabschätzung in Ökobilanzen. Öbu SR 28/2008. Zürich, Öbu, BAFU**:** 194.

Galiasso-Tailleur, R. (2007). "Effect of recycling the unconverted residue on a hydrocracking catalyst operating in an ebullated bed reactor." Fuel Processing Technology **88**: 779-785.

Ganapathya, T., R. P. Gakkharb and K. Murugesan (2011). "Influence of injection timing on performance, combustion and emission characteristics of Jatropha biodiesel engine." Applied Energy **88**(12): 4376-4386.

Garnier-Laplace, J. C., K. Beaugelin-Seiller, R. Gilbin, C. Della-Vedova, O. Jolliet and J. Payet (2008). A Screening Level Ecological Risk Assessment and ranking method for liquid radioactive and chemical mixtures released by nuclear facilities under normal operating conditions. Proceedings of the International conference on radioecology and environmental protection, 15-20 june 2008, Bergen. International conference on radioecology and environmental protection. Bergen.

Gassner, M., F. Vogel, G. Heyen and F. Maréchal (2011). "Process design of SNG production by hydrothermal gasification of waste biomass: Thermoeconomic process modelling and integration." Energy & Environmental Science **4**: 1726-1741.

GEXSI (2008). Global market study on Jatropha - final report. Berlin, GEXSI LLP**:** 187.

Gmünder, S., C. Naranjo, C. Ortiz, R. Zah and M. Gauch (2012). First chapter of Final Report: Life Cycle Assessment. Proyecto: "Estrategias de energía sostenible y biocombustibles para Colombia ATN/JC-10826-CO y ATN/JF-10827-CO" "Evaluación del ciclo de vida de la cadena de producción de bio-combustibles en Colombia". PREPARADO PARA: Banco Interamericano de Desarrollo (BID) –Ministerio de Minas y energía.

Gmünder, S., R. Singh, S. Pfister, A. Adheloya and R. Zah (in press). "Environmental impacts of Jatropha curcas biodiesel in India." Journal of Biomedicine and Biotechnology - special issue Biofuels 2012.

Goedkoop, M., R. Heijungs, M. Huijbregts, A. D. Schryver, J. Struijs and R. v. Zelm (2009). ReCiPe 2008: A life cycle impact assessment method which comprises harmonised category indicators at the midpoint and the endpoint level. First edition. Report I:

Characterisation. R. e. Milieu. Den Haag, NL, Ministry of Housing, Spatial Planning and the Environment (VROM).

Goedkoop, M. and R. Spriensma (2001). *The Eco-Indicator 99: a DamageOriented Method for Life Cycle Impact Assessment: Methodology Report*. Publikatiereeks produktenbeleid; nr. 36A. Den Haag, Ministerie van Volkshiusvesting, Ruimtelijke Ordening en Milieubeheer.

Graboski, M. S., R. L. McCormick, T. L. Alleman and A. M. Herring (2003). The Effect of Biodiesel Composition on Engine Emissions from a DDC Series 60 Diesel Engine. Golden, Colorado, Colorado Institute for Fuels and Engine Research

Grieg-Gran, M., M. Haase, J. J. Kessler, S. Vermeulen and E. Wakker (2007). The Dutch economic contribution to worldwide deforestation and forest degradation. London, Report prepared for Greenpeace Netherlands**:** 113.

Hanumantha Rao, Y. V., R. S. Voleti, V. S. Hariharan and A. V. Sitarama Raju (2008). "Jatropha oil methyl ester and its blends used as an alternative fuel in diesel engine." Int J Agric & Biol Eng **1**(2): 32-38.

Heck, T. (2007). Wärme-Kraft-Kopplung. Sachbilanzen von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. Final report ecoinvent No. 6-XIV. R. Dones. Dübendorf & Villigen, CH, Swiss Centre for Life Cycle Inventories & Paul Scherrer Institut

Helin, M. (2005). Moisture in wood fuels and drying of wood chips, North Karelia Polytechnic.

Heller, J. (1997). J. curcas: potential: limitations and future research needs. Biofuels and Industrial Products from Jatropha Curcas. M. M. G.M. Gübitz, M. Trabi. Graz DBV**:** 242–254.

Hersener, J.-L., E. Meister, V. Mediavilla, A. Lips, J. Rüegg, T. Nussbaumer, U. Baserga, D. Müller, F. Dinkel, B. Waldeck, T. Trauboth Müller and B. Hirs (1997). Schlussbericht: Projekt Energiegras/Feldholz. Bern, CH, Bundesamt für Energiewirtschaft.

Hölscher, T., S. Deimling, R. Freiermuth Knuchel, G. Gaillard, T. Kägi and K. Müller-Sämann (2007). Energieproduktion aus landwirtschaftlicher Biomasse am Oberrhein – Auswirkungen für Landwirtschaft und Umwelt. Endbericht zum Projekt Überprüfung der Möglichkeiten einer landwirtschaftlichen Erzeugung von Bioenergieträgern als Alternative zu intensiven Ackerbaukulturen mit dem Ziel der Nitratsanierung des Grundwassers., Agentur für Nachhaltige Nutzung von Agrarlandschaften (ANNA) Müllheim, Forschungsanstalt Agroscope Reckenholz-Tänikon ART, PE INTERNATIONAL GmbH**:** 69 p.

Houghton, J. T., L. G. Meira Filho, B. Lim, K. Treanton, I. Mamaty, Y. Bonduki, D. J. Griggs and B. A. Callender (1996). Revised 1996 IPCC Guidelines for National Greenhouse Gas Inventories. . Bracknell, UK Meteorological Office.

Houghton, R. A., D. L. Skole, C. A. Nobre, J. L. Hackler, L. K.T. and C. W.H. (2000). "Annual fluxes of carbon from deforestation and regrowth in the Brazilian Amazon." Nature **403**: 301-304.

Huang, J., Y. Wang, J. Qin and A. P. Roskilly (2010). "Comparative study of performance and emissions of a diesel engine using Chinese pistache and jatropha biodiesel." Fuel Processing Technology **91**: 1761-1767.

Huijbregts, M. and R. v. Zelm (2012). Uncertainty in life cycle impact assessment: typologies, tools and a case of ecotoxicity. LC impact workshop, 20.01.2012. ETHZ, Zürich.

IPCC (2001). Climate Change 2001: The Scientific Basis Third Assessment Report of the Intergovernmental Panel on Climate Change (IPCC). J. T. Houghton, et al. (eds). Cambridge, UK, IPCC, Intergovernmental Panel on Climate Change, Cambridge University Press, The Edinburgh Building Shaftesbury Road.

IPCC (2006). Guidelines for national greenhouse gas inventories. Volume 4: Agriculture, forestry and other land use. L. B. Simon Eggleston, Kyoko Miwa, Todd Ngara and Kiyoto Tanabe. Kanagawa.

IPCC (2007). IPCC Climate Change Fourth Assessment Report: Climate Change 2007. http://www.ipcc.ch/ipccreports/assessments-reports.htm.

Jayalakshmi, S., V. Sukumaran and J. Kurian (2007). Hydrogen Production from Kitchen Waste using Heat Treated Anaerobic Biogas Plant Slurry. Proceedings of the International Conference on Sustainable Solid Waste Management, 5 - 7 September 2007, Chennai, India. pp.356-362. http://www.swlf.ait.ac.th/IntlConf/Data/ICSSWM%20web/FullPaper/Session%20VI-B/6_B2%20_Jayalakshmi_.pdf (9. Nov 2010). International Conference on Sustainable Solid Waste Management, 5 - 7 September 2007, Chennai, India.

Jindal, S., B. P. Nandwana and N. S. Rathore (2010). "Comparative Evaluation of Combustion, Performance, and Emissions of Jatropha Methyl Ester and Karanj Methyl Ester in a Direct Injection Diesel Engine." Energy Fuels **24**: 1565-1572.

Jingura, R. M., D. Musademba and R. Matengaifa (2010). "An evaluation of utility of Jatropha curcas L. as a source of multiple energy carriers." International Journal of Engineering, Science and Technology **2**(7): 115-122.

Jongschaap, R. E. E., W. J. Corré, P. S. Bindraban and W. A. Brandenburg (2007). Claims and Facts on Jatropha curcas L. Report 158. Wageningen, Plant Research Institut.

Jungbluth, N. (2007). Erdöl. Sachbilanzen von Energiesystemen: Grundlagen für den ökologischen Vergleich von Energiesystemen und den Einbezug von Energiesystemen in Ökobilanzen für die Schweiz. ecoinvent report No. 6-IV. R. Dones. Duebendorf, CH, Swiss Centre for Life Cycle Inventories.

Jungbluth, N., M. Chudacoff, A. Dauriat, F. Dinkel, G. Doka, M. Faist Emmenegger, E. Gnansounou, N. Kljun, K. Schleiss, M. Spielmann, C. Stettler and J. Sutter (2007). Life Cycle Inventories of Bioenergy. ecoinvent report No. 17. Dübendorf, CH, Swiss Centre for Life Cycle Inventories.

Kaiser, E. R. (1966). Chemical Analyses of Refuse Components. Department of Chemical Engineering, New York University. The American Society of Mechanical Engineers ASME, National Incinerator Conference 1-4 May 1966. Conference Proceedings, p.84-88. http://www.seas.columbia.edu/earth/wtert/sofos/nawtec/1966-National-Incinerator-Conference/1966-National-Incinerator-Conference-08.pdf (16. Nov 2010). National Incinerator Conference 1-4 May 1966.

Kaiser, E. R. (1975). Physical-Chemical Character of Municipal Refuse. Proc. 1975 Int. Symp on energy Recovery from Refuse, University of Louisvile, Louisvile, KY (1975) Int. Symp on energy Recovery from Refuse, University of Louisvile, Louisvile, KY.

Keesom, W. and J. Moretta (2009). Life cycle Assessment of North American and Imported Crudes. Chicago, Jacobs Consultancy.

Kinast, J. A. (2003). Production of biodiesels from multiple feedstocks and properties of biodiesels and biodiesel/diesel blends. Final report. Report 1 in a series of 6. Subcontractor report. Golden, Colorado, NREL, National Renewable Energy Laboratory.

Koellner, T., L. de Baan, T. Beck, M. Brandao, B. Civit, M. Goedkoop, M. Margni, L. i Canals, R. Müller-Wenk, B. Weidema and B. Wittstock (2012). "Principles for life cycle inventories of land use on a global scale." The International Journal of Life Cycle Assessment: 1-13.

Kranert, M., R. Gottschall, C. Bruns, G. Hafner, O. Schiere, C. Seibel, M. Dohmann, K. Fricke, M. Fricke, A. Karwatzki, E. Kraft, J. Lippross, K. Opphard, T. Pretz, G. Rettenberger, H. Seier, R. Widmann, R. Schäfer, A. Ochs and M. Weyers (2007). Grünabfälle - besser kompostieren oder energetisch verwerten? - Vergleich unter den Aspekten der CO2-Bilanz und der Torfsubstitution. EdDE-Dokumentation Nr. 11. Entsorgergemeinschaft der Deutschen Entsorgungswirtschaft e.V., Köln, Germany http://www.entsorgergemeinschaft.de/index.php?eID=tx_nawsecuredl&u=0&file=uploads/tx_skpdfviewer/EdDE-Doku-11.pdf&t=1289369757&hash=ce70dcbe04ed51c4ac6b996152ada497 (9. Nov 2010). Köln, Entsorgergemeinschaft der Deutschen Entsorgungswirtschaft e.V.

Löschau, M. (2009). Abfallverbrennung als Emissionsquelle – Der Beitrag der thermischen Abfallbehandlung zur gesamten Schadstoffemission in Deutschland ist gering. ReSource No 4, 2009, p.30-37. http://www.itad.de/media/www.itad.de/org/med_90080/315_loeschau_margit_abfallverbrennung_als_emissionsquelle_in_resource_42009_s._30-37.pdf (25 Feb 2011).

Luterbacher, J., M. Fröling, F. Vogel, F. Maréchal and J. W. Tester (2009). "Hydrothermal Gasification of Waste Biomass: Process Design and Life Cycle Assessment." Environ. Sci. Technol. **43**: 1578-1583.

Macedo, M. N., R. S. DeFries, D. C. Morton, C. M. Stickler, G. L. Galford and Y. E. Shimabukuro (2012). "Decoupling of deforestation and soy production in the southern Amazon during the late 2000s." Proceedings of the National Academy of Sciences **109**(4): 1341-1346.

Madejón, E., R. López, J. M. Murillo and F. Cabrera (2001). "Agricultural use of three (sugar-beet) vinasse composts: effect on crops and chemical properties of a Cambisol soil in the Guadalquivir river valley (SW Spain)." Agriculture, Ecosystems & Environment **84**(1): 55-65.

Mark, F. E. and R. Martin (1994). The Role of Plastics in Municipal Solid Waste Combustion. technical paper No. 8004 of APME (Association of Plastics Manufacturers in Europe), Brussels. Brussels, APME (Association of Plastics Manufacturers in Europe).

Martini, G., C. Astorga, T. Adam, A. Farfaletti, U. Manfredi, L. Montero, A. Krasenbrink, B. Larsen and G. De Santi (2009). Effect of Fuel Ethanol Content on Exhaust Emissions of a Flexible Fuel Vehicle. Ispra, Transport & Air Quality Unit, EC-Joint Research Centre.

Maystre, L.-Y., V. Duflon, T. Diserens, D. Leroy, J. Simos and F. Viret (1994). Déchets urbains – Nature et caractérisation. Lausanne Presses Po¬litech¬niques et Universitaires Romandes.

McKnight, C., W. Schiewe, J. M. Ginestra, W. Ijlstra, D. Rauschning and D. E. Sherwood (2007). Joint Ebullated Bed Catalyst Development Program Results in Over 30% Reduction in Catalyst Usage. Houston, US, Criterion Catalysts & Technologies.

Menzi, H., R. Frick and R. Kaufmann (1997). Ammoniak-Emissionen in der Schweiz: Ausmass und technische Beurteilung des Reduktionspotentials. Schriftenreihe der FAL 26. Zürich-Reckenholz.

Meraz, L., M. Oropeza and A. Dominguez (2002). "Prediction of the Combustion Enthalpy of Municipal Solid Waste. http://www.springerlink.com/content/u5101w24k5676078/fulltext.pdf (9. Nov 2010)." Chem. Educator **7**: 66-70.

Milà i Canals, L., G. Rigarlsford and S. Sim (2012). "Land use impact assessment of margarine." The International Journal of Life Cycle Assessment: 1-13.

Milà i Canals, L., J. Romanyà and S. J. Cowell (2007). "Method for assessing impacts on life support functions (LSF) related to the use of 'fertile land' in Life Cycle Assessment (LCA)." Journal of Cleaner Production **15**(15): 1426-1440.

Mogaka, V., M. Iiyama, AlbrechtEhrensperger, M. Birtel and S. Gmuender (in preparation). "Understanding adoption of Jatropha curcas L. by smallholders in Kenya: The examples of Bondo, Kibwezi, and Kwale districts."

MT-Biomethan (2011) "Amine Washing. Gas refinement technology – process description (Amine washing)." **Volume**, DOI:

NBB (2012). Biodiesel Fact Sheets: Emissions, National Biodiesel Board, US.

Nemecek, T., O. Huguenin-Elie, D. Dubois and G. Gaillard (2005). Ökobilanzierung von Anbausystemen im schweizerischen Acker- und Futterbau. Schriftenreihe der FAL. Reckenholz, Zürich, Agroscope FAL 155 p.

Nemecek, T., T. Kägi and S. Blaser (2007). Life Cycle Inventories of Swiss and European Agricultural Production Systems. Final report ecoinvent V2.0 No. 15a. , Agroscope

Reckenholz-Taenikon Research Station ART, Swiss Centre for Life Cycle Inventories, Zurich and Dübendorf, CH.

Newman, R. (2003). MISCANTHUS Practical Aspects Of Biofuel Development. London, UK, Department of Trade & Industry.

NRC (2008). GHGenius, Natural Resources Canada, Office of Energy Efficiency.

Ntziachristos, L. and P. Boulter (2009b). EMEP/EEA air pollutant emissions inventory guidebook 2009: Road vehicle tyre and brake wear; Road surface wear. Copenhagen, European Environment Agency.

Ntziachristos, L., Z. Samaras, C. Kouridis, D. Hassel, I. McCrae, J. Hickman, K.-H. Zierock, M. Keller, M. Andre, N. Gorissen, P. Boulter, R. Joumard, R. Rijkeboer, S. Geivanidis and S. Hausberger (2009a). EMEP/EEA air pollutant emissions inventory guidebook 2009: Exhaust emissions from road transport. Copenhagen, European Environment Agency.

Nussbaumer, T. (2006). Energietechnik und Umwelt. Vorlesungsskript ETH Zürich, 15. Auflage 2006, ISBN 3-908705-05-3. .

Ogunwole, J. O., D. R. Chaudhary, A. Ghosh, C. K. Daudu, J. Chikara and J. S. Patolia (2008). "Contribution of Jatropha curcas to soil quality improvement in a degraded Indian entisol." Acta Agriculturae Scandinavica, Section B - Soil & Plant Science **58**(3): 245-251.

Oil-Sands-Review (2009). Oil Sands Bitumen Production: Production by project. Available by trial or on subscription. Available at: http://www.oilsandsreview.com/statistics/production.asp.

Ordorica-Garcia, G., E. Croiset, P. Douglas, A. Elkamel and M. Gupta (2007). "Modeling the Energy Demands and Greenhouse Gas Emissions of the Canadian Oil Sands Industry." Energy & Fuels **21**: 2098-2111.

Page, S. E., R. Morrison, C. Malins, A. Hooijer, J. O. Rieley and J. Jauhiainen (2011). Review of peat surface greenhouse gas emissions from oil palm plantations in Southeast Asia. ICCT White Paper 15. Washington, International Council on Clean Transportation.

Posch, M., J. Seppälä, J. P. Hettelingh, M. Johansson, M. Margni and O. Jolliet (2008). "The role of atmospheric dispersion models and ecosystem sensitivity in the determination of characterisation factors for acidifying and eutrophying emissions in LCIA." International Journal of Life Cycle Assessment **13**: 477–486.

Prasuhn, V. and E. Spiess (2003). Regional differenzierte Abschätzung der Nitratauswaschung über Betriebszählungsdaten. 10. Gumpensteiner Lysimetertagung, 29. und 30. April 2003, Bundesanstalt für alpenländische Landwirtschaft Gumpenstein, A-8952 Irdning.

Rabl, A. and J. V. Spadaro (2004). The RiskPoll software, version is 1.051 (dated August 2004). www.arirabl.com.

Reinhard, J. (2007). Influence of Biofuel production to Food production and the consequential environmental Impacts for Switzerland:

Consequential Life Cycle Assessment (CLCA). Berlin, FHTW/EMPA. **Master Thesis**.

Reinhardt, G., N. Rettenmaier, S. Gärtner and A. Pastowski (2007). Regenwald für Biodiesel? Franfurt am Main, WWF.

Reisinger, K., C. Haslinger, M. Herger and H. Hofbauer (1996). BIOBIB - A database for biofuels. THERMIE Conference: Renewable Energy Databases. Harwell, UK and Vienna., Institute of Chemical Engineering and Fuel and Environmental Technology, University of Technology.

Richner, W., H. R. Oberholzer, F. Knuchel, H. R., O., S. Ott and U. Walther (in prep.). Modell zur Beurteilung der Nitratauswaschung in Ökobilanzen – SALCA-NO3. To be published at: www.agroscope.ch, Agroscope Reckenholz - Tänikon ART**:** 27p. .

Rolland, C., Scheibengraf, M. (2003). Biologisch Abbaubarer Kohlenstoff Im Restmüll. http://www.ubavie.gv.at/fileadmin/site/publikationen/BE236.pdf (9. Nov 2010). Berichte BE-236. Vienna, Austria, Umweltbundesamt GmbH.

Rosenbaum, R. K., T. M. Bachmann, L. S. Gold, M. A. J. Huijbregts, O. Jolliet, R. Juraske, A. Köhler, H. F. Larsen, M. MacLeod, M. Margni, T. E. McKone, J. Payet, M. Schuhmacher, D. van de Meent and M. Z. Hauschild (2008). "USEtox - The UNEP SETAC toxicity model: recommended characterisation factors for human toxicity and freshwater ecotoxicity in Life Cycle Impact assessment." International Journal of Life Cycle Assessment **13**(7): 532-546.

Roy, R. N., R. V. Misra, J. P. Lesschen and E. M. Smaling (2003). Assessment of soil nutrient balance: Approaches and Methodologies. Rome, FAO, FOOD AND AGRICULTURE ORGANIZATION OF THE UNITED NATIONS.

Sales, A. and S. A. Lima (2010). "Use of Brazilian sugarcane bagasse ash in concrete as sand replacement." Waste Management **30**(6): 1114-1122.

Schleiss, K. and W. Edelmann (2000). Stromproduktion aus Feststoff-Vergärung – Ökobilanz . Commissioned by Biogas Forum and Bundesamt für Energie (Swiss Federal Office of Energy) Projekt Nr.: 30 703. http://www.energieforschung.ch/ENET/Publication/ENETProj.nsf/C69C83D4B9E14D9FC1256A3B0052E62E/$FILE/200119.pdf (9. Nov 2010). Bern, CH, Biogas Forum and Bundesamt für Energie.

Schmid, M., A. Neftel and J. Fuhrer (2000). Lachgasemissionen aus der Schweizer Landwirtschaft. Schriftenreihe der FAL 33**:** 131 p.

Schmidt, J. H. (2008). "System delimitation in agricultural consequential LCA: outline of methodology and illustrative case study of wheat in Denmark." Int J LCA **13**: 350-364.

Scurlock, J. M. O. (1999). Miscanthus: A Review of European Experience with a Novel Energy Crop. Oak Ridge, USA, Oak Ridge National Laboratory.

Seppälä, J., M. Posch, M. Johansson and J. P. Hettelingh (2006). "Country-dependent Characterisation Factors for Acidification and Terrestrial Eutrophication Based on Accumulated Exceedance as an Impact Category Indicator." International Journal of Life Cycle Assessment **11**(6): 403-416.

Shell-Canada (2009). Shell and the Canadian Oil Sands. 2009 Factbook.

Sleeswijk, A. W., L. F. C. M. van Oers, J. B. Guinée, J. Struijs and M. A. J. Huijbregts (2008). "Normalisation in product life cycle assessment: An LCA of the global and European economic systems in the year 2000." Science of The Total Environment **390**(1): 227-240.

Soobadar, A. and K. F. R. Ng Kee Kwong (2011). "Impact of High Rates of Vinasse on Some Pertinent Soil Characteristics and on Sugarcane Yield in Mauritius." Journal of Sustainable Agriculture **36**(1): 36-53.

Spielmann, M., C. Bauer, R. Dones and M. Tuchschmid (2007). Transport Services. ecoinvent report No. 14. Dübendorf, CH, Swiss Centre for Life Cycle Inventories.

Stauffer, W., V. Prasuhn and E. Spiess (2001). Einfluss unterschiedlicher Fruchtfolgen auf die Nitratauswaschung. Neue Erkenntnisse zu Stickstoffflüssen im Ackerbau. FAL-Tagung 6.4.2001.

Surbeck, A. and D. Leu (1998). Toxikologische Beurteilung von Nitrat im Trinkwasser. Region Klettgau - Verantwortungsvoller Umgang mit Boden. R. W. Scholz, Bösch, S., Mieg, H.A., Stünzi, J. (Eds). Zürich, ETH-UNS-Fallstudie '97, Verlag Rüegger**:** 203-205.

SVGW (2004). Richtlinien für die Einspeisung von Biogas ins Erdgasverteilnetz. . Bern, CH, Swiss Gas and Water Industry Association.

T&M-Leuven (2007). TREMOVE v2.7b, Transport & Mobility Leuven. Brussels, European Commission.

Thitame, S. N., G. M. Pondhe and D. C. Meshram (2010). "Characterisation and composition of Municipal Solid Waste (MSW) generated in Sangamner City, District Ahmednagar,

Maharashtra, India. http://www.springerlink.com/content/86326l5271002585/fulltext.pdf (9. Nov 2010)." Environ Monit Assess **170**: 1–5.

Thöni, L., E. Seitler and D. Matthaei (2007). Ammoniak-Immissionsmessungen in der Schweiz 2000 bis 2006, im Auftrag des Bundesamtes für Umwelt BAFU, der OSTLUFT un der Kantone Luzern und Freiburg, available at: http://www.bafu.admin.ch/luft/00649/01960/index.html?lang=de.

Tillman, D. A. (1991). Combustion of Solid Fuels and Wastes. New York Academic Press.

Van-Loo, S. and J. Koppejan, Eds. (2008). The Handbook of Biomass Combustion and Co-firing. London, Earthscan.

Van Oers, L., A. De Koning, J. B. Guinee and G. Huppes (2002). Abiotic Resource Depletion in LCA.; Improving Characterization Factors Abiotic Resource Depletion as Recommended in the new Dutch LCA Handbook. RWS-DWW report.

Van Zelm, R., M. A. J. Huijbregts, H. A. Den Hollander, H. A. Van Jaarsveld, F. J. Sauter, J. Struijs, H. J. Van Wijnen and D. Van de Meent (2008). "European characterization factors for human health damage of PM10 and ozone in life cycle impact assessment." Atmospheric Environment **42**: 441-453.

Walther, U., J.-P. Ryser and R. Flisch (2001). "Grundlagen für die Düngung im Acker- und Futterbau." Agrarforschung **8**(6): 1-80.

Weidema, B. P., C. Bauer, R. Hischier, C. Mutel, N. T., J. Reinhard, C. O. Vadenbo and G. Wernet (2012). Overview and methodology. Data quality guideline for the ecoinvent database version 3 (final draft; revision 2). ecoinvent report No. 1(v3). St. Gallen, Ecoinvent Centre Swiss Centre for Life Cycle Inventories.

Wellinger, A., W. Edelmann, M. Schmid, J. Wochele and H.-C. Angele (2006). Energieproduktion aus Küchenabfällen - Ein Vergleich der Vergärung mit der Verbrennung in KVA. http://www.biogasmax.eu/media/3_energieproduktion__037083000_1208_19042007.pdf (9. Nov 2010) and http://www.biomasseenergie.ch/dt/pdf/Studie_Verg%E4rung_KVA_231006.pdf (28 Oct 2006). Aadorf, Switzerland, BiomasseSchweiz.

Wicke, B., V. Dornburg, M. Junginger and A. Faaij (2008). "Different palm oil production systems for energy purposes and their greenhouse gas implications." Biomass and Bioenergy **32**(12): 1322-1337.

Williams, R. B., B. M. Jenkins and D. Nguyen (2003). Solid Waste Conversion: A review and database of current and emerging technologies. December, 2003. http://biomass.ucdavis.edu/materials/reports%20and%20publications/2003/2003_Solid_Waste_Conversion.pdf (9. Nov 2010). Davis, California, Department of Biological and Agricultural Engineering, University of California.

WMO (1999). Scientific Assessment of Ozone Depletion: 1998. Global Ozone Research and Monitoring Project - Report No. 44, ISBN 92-807-1722-7. Geneva.

Yang, Y., J. Bae, J. Kim and S. Suh (2012). "Replacing Gasoline with Corn Ethanol Results in Significant Environmental Problem-Shifting." Environmental Science & Technology.

Zah, R., H. Böni, M. Gauch, R. Hischier, M. Lehmann and P. W. (Empa) (2007). Ökobilanz von Energieprodukten: Ökologische Bewertung von Biotreibstoffen. Bern, BfE/BAFU/BLW.

Zimmermann, P., G. Doka, F. Huber, A. Labhardt and M. Menard (1996). Ökoinventare von Entsorgungsprozessen, Grundlagen zur Integration der Entsorgung in Ökobilanzen. ESU-Reihe, 1/96. Zürich, Institut für Energietechnik, ETH Zürich.