Phytoremediation: a plant-based alternative for conventional remediation technologies for sustainable use and management of metal contaminated sandy agricultural land.

Case study: the Campine (BE)

Proefschrift voorgelegd tot het behalen van de graad van
Doctor in de Wetenschappen,

Nele Witters

Promotor:
Prof. dr. Jaco Vangronsveld

Co-promotor:
Prof. dr. Steven Van Passel

Commissieleden:
Prof. dr. Jaco Vangronsveld
Prof. dr. Steven Van Passel
Prof. dr. Jan Yperman

ACKNOWLEDGEMENTS

Theo Thewys, thank you so much for the life changing opportunity you gave me 4 years ago to start a dissertation in this emerging field. I will forever be grateful for the chances I have been given through this research. And not only did you give me your trust, you were also showing this to other people, to make it possible for me to build up a network which is so important nowadays. Thank you for your never ending patience with my stubborn character, for keeping me on track. It has been a bumpy ride, but those are the ones you remember...

Jaco Vangronsveld, thank you for your endless patience in editing manuscripts, for your calm, for your 24/7 service, for helping me to build up a network, for sharing your knowledge, for your constant support, for your confidence, … Thank you, a lot.

Bernard Vanheusden, every once in a while you would send me interesting articles, conferences, projects, and lectures. It is nice to know that there are people that are genuinely interested in what you do. Also, you were a very thorough reviser, and a good mentor. I was hugely inspired by your never ending enthusiasm.

Steven Van Passel, as a young academic in the same field of research I am impressed by your managing skills. You guided me through the turmoil of European projects. Also, you were never judgmental and ever supportive.

Robert Mendelsohn, thank you for giving me the opportunity to work in such an inspiring environment as the School of Forestry & Environmental Studies. I was challenged every day. And that’s what I need to keep on going. Thank you for your interesting comments, for your little hints, for pushing my limits so that I would go home again and rethink my whole argumentation. Thank you also for giving me the opportunity and your trust to organize my very first conference at F&ES. We even had a full house! And nice pizza afterwards…

Mijn collega’s, de F13, my American colleagues, we are all in the same position. Working hard. Complaining. Discussing. The stress for deadlines. The rush when a paper is accepted. But in the end we would all have to agree: we wouldn’t want another job!

Ann Peters, Nele Nivelle, thank you for translating my sometimes unrealistic enthusiasm for new challenges into real life opportunities. It is mostly your work, your enthusiasm, and your trust that made it possible for me to go the

United States with an FWO scholarship for one year. You have no idea what an impact this experience has on my dissertation and life, what an impact you have made. I'm pretty sure you will find me in your office more often…
Magda Ieven, you've got some nice graph and picture editing skills. If it weren't for you, this thesis would be stained with 'paint' edited graphs. Thank you, you saved me a lot of time and stress and saved the readers a lot of time figuring out what I'm trying to represent in my graphs and figures. You took care of the finishing touch of this paper.
Ann Ruttens, you were there for me in the first years. I remember how I came to your office a couple of times and asked for the most impossible detailed data. And you were so patient. You were also the one who helped me pick my first conference and write my abstract. I'm a fan.

Stijn Van Slycken. How to thank you? If it weren't for you, this thesis you're about to read might not have existed. You were there for me not only to support me as a colleague, but you very soon became a close (even with 5,000 miles between us) friend. I am sure we will work together in the future, this tandem works well.

When I am not gliding through the air in a plane, enjoying the beauty of what I am seeing down below (this is not a metaphor), I have friends that keep me with both feet on the ground (this is), and assure me that a scientist can try to become immortal on paper, but, as a human being, still needs food and drinks.

Valery-Ann: By now, after listening for me for 4 years, I guess you know as much about the economics of phytoremediation as I do. You never showed the least boredom, and I think you actually understood what I was trying to explain. Not bad for a master in astronomy…
Marcoen, my dearest friend, my diary. Thanks for introducing me to Wulffmorgenthaler. They are my favorite lunch break, besides emailing you 4 page long emails every two weeks, venting and talking about absurd and completely unnecessary stuff.

Jutta: You've known me for 15 years now and all that time I was studying, working and complaining. I have to tell you, you've got some bad taste in friends. But I like you!

Hanna: My favorite German. It's been wonderful and crazy knowing you! What else is there to say? Setting deadlines, motivating each other to get up at 6:30 on Saturdays for a quick coffee and then start working in Kroon. You were there for me in the most exciting and turbulent period of my life. Within 40 years I hope we will be rocking our chairs together on some nice porch. For now, we have Skype and cheap flights.

Rakina: I never met a person as smart as you, post-doc at 27! I still haven't figured out exactly what you're doing with the enzymes, but one day you'll save the world. Your never ending enthusiastic 'No worries' and 'Everything will work out' in your most funniest Bulgarian accent will leave a permanent mark in my ear. As well as your hilarious interpretation of Hotel California by the way... I already miss our nightly walks for a coffee to 'just relax for an hour' and then go back to work, or our Sunday coffees at 8 in café Romeo while our laundry was being done. See you in San Francisco!

Ellen: You are by far the kindest person I have ever met. Thank you for introducing me to the American culture (what to say, and most importantly what not to say to people). Thank you for keeping me healthy with your nice meals and desserts. I'm looking forward to a next board game night. For everything, arigatô (please don't make me pronounce this).

How do you start thanking people who put their lives on hold for you?

Mama, dankuwel voor je bezorgdheid. Gedurende mijn hele studiecarrière, heb je me altijd geholpen met mijn studies, gaande van het uitbeelden van een zon, over het regelen van je werkuren, tot het gewoon er zijn voor mij (liefst met chocolade cake). *Papa*, dankuwel voor je kalme genen die ik van jou geërfd heb, en ook om mama te kalmeren wanneer zij meer gestresst was dan ik. Dankuwel ook voor je ongelooflijke vertrouwen in mij en om mij steeds opnieuw te volgen in mijn zotte enthousiasme. *Katrien*, kleine meid, mijn manusje van alles, we moeten er dringend tussenuit. *Koen*, je hebt me vaak geambeteerd met je vragen hoe het nu precies zat, of ik nu eens precies wilde uitleggen wat ik deed en of ik nu nog niet klaar was na hoeveel jaar was het ook al weer? Je bent een superbroer.

Mijn familie: dit is niet zomaar een beleefdheidvermelding. Ik ben gezegend met een hechte familie die ongelooflijk bezorgd zijn om elkaar en oprechte interesse hebben in elkaars idealen. Dit is weer maar eens gebleken tijdens mijn verblijf in de VS: dankjewel voor de mailtjes, kaartjes, telefoontjes, berichtjes.

Johan: We zijn een goed team. En we hebben het gehaald! Iets zegt me dat het nu pas begint…

Nele Witters
March 2011

SUMMARY

When, due to technical or financial reasons, conventional remediation technologies for soil are inapplicable, literature often suggests phytoremediation as an alternative remediation technology. Moreover, using biomass originating from contaminated land as a feedstock for energy production or for (chemical) industry may contribute to the reduction of carbon dioxide (CO_2) emissions, compared to conventional remediation technologies, but also within the wider framework of the European Renewable Energy Directive. However, the harvested biomass may contain increased amounts of metals and these need safe treatment to avoid new spreading in the environment and subsequent health costs. The integration of all these aspects of phytoremediation in an economic study has not been performed before.

Phytoremediation undoubtedly has a high potential to enhance the degradation and/or removal of organic contaminants from soils and undeep groundwater (Vangronsveld *et al.*, 2009; Weyens *et al.*, 2009, 2010). However, based on extrapolations of data obtained from pot experiments, enthusiastic promises have been made concerning the potential of metal phytoextraction (Salt *et al*., 1995; Rulkens *et al.*, 1998; Susarla *et al*., 2002; Mench *et al.*, 2010). In general, a trace element phytoextraction protocol consists of the following elements (Vangronsveld *et al.*, 2009):

(i) cultivation of the appropriate plant/crop species on the contaminated site;

(ii) removal of the harvestable trace element enriched biomass from the site; and

(iii) post-harvest treatment of the biomass (*i.e.* digestion, pressing, combustion).

Regarding (i), a key question which has been in debate since the very beginning of the introduction of the trace element phytoextraction concept is: "Should one use trace element hyperaccumulator plants or high biomass producing plants?" Regarding (ii) and (iii), the main barriers to the development of commercially viable phytoextraction procedures for trace elements remain the long time required to remediate soil to threshold values, and the use/valorization/disposal of the contaminated biomass.

Each incidence of pollution is different and successful sustainable management requires the careful integration of all relevant factors. More data are needed to quantify the underlying economics, and more demonstration projects on metal extraction are required to provide recommendations and convince regulators, decision makers, and the public of the applicability of phytotechnologies as an alternative for conventional remediation technologies for the treatment of soils, brownfields, groundwater, and wastewater contaminated with toxic metals (and organic pollutants as well) (Ruttens and Vangronsveld, 2006; Adriaensen *et al.,* 2008; Vangronsveld *et al.*, 2009). Therefore, a clear road map for utilization of phytotechnologies needs to be developed to allow the user to make an all things considered decision on whether or not to apply phytotechnologies, and if so, on the most suitable crop-conversion technology (Adriaensen *et al.*, 2008). We developed an optimization tool for the assessment of different crop-conversion methodologies, based on a moderately metal enriched agricultural soil in the Campine region (700 km² in Belgium and the Netherlands). The strategic aim for this region is the restoration of Cd contaminated soils for conventional agricultural use by means of crops that remediate the soil, while generating an income.

In **Section 1** we discuss the background conditions of this Campine case study, as well as current federal and regional legislation on soil remediation, and we provide an economic analysis of policy development for environmental or health purposes. We focus our analysis on three (energy) crops: energy maize (*Zea mays*), rapeseed (*Brassica napus*), and short rotation coppice of willow (*Salix spp.*) based on current experience with these crops and the availability of data from the experimental field (Lommel) in the Campine region.

Elevated metal concentrations in soils lead to increased metal concentrations in plants. A crucial aspect when growing crops on contaminated soil is the fact whether the harvested crops will be classified as (hazardous) waste, or as biomass since this has an impact on the further utilization and valorization of the crop. Therefore, before developing a decision model on phytoremediation crops, in **Section 3** we unravel the multitude of definitions and regulations that are related to the concept and use of metal enriched energy crops, as this might have technical and thus economic implications for conversion processes. We found no evidence of legislation that could exclude energy crops grown on metal contaminated land from being classified as biomass. We suggest that classification of energy crops with metal accumulating purposes should depend on the main purpose of growing them. If the main purpose is sustainable use of contaminated agricultural land in function of energy production, crops should be considered as biomass. Phytoremediation can be denominated as the main purpose when it involves crops which are capable of accumulating metals such that concentrations in soil decrease substantially (such as with hyperaccumulators). Given the relatively low accumulation potential of the crops investigated in this work, we consider sustainable land management with energy production for alternative income generation as the main purpose. For wood, the distinction between biomass and waste might actually become important for conversion, as different legislation will apply depending on classification, having implications on types of combustion installations and emission control. For energy maize and rapeseed, there is no explicit legislation on input for energy conversion installations, but there exist threshold values for the rest product.

For the three crops considered in our case study we found that metals even have no marginal effect on the energy conversion (technical) efficiency. Moreover, no metals end up in the energy carrier, but they get concentrated in the rest products: ashes, digestate, and cake. Unsafe use and disposal might lead to substantial risks for human health and the environment in general. The presence of toxic metals in the residue after conversion of biomass is a scholarly example of an externality as this fact is not reflected in the price of the original biomass. We study the impact of the contamination on processing the rest products for secondary use or disposal, since, to comply with legislation, a different end use might have to be found for (part of) the rest products. Strictly applying current legislation, applying part of the digestate on the land and combusting the rest gives the highest total economic value. For cake, the use as fodder results in no marginal impact from metals. Willow ashes have to be disposed off because metal concentrations are too high. The difference in economic value between contaminated and uncontaminated rest product is taken into account in the final optimization model.

Performing a Life Cycle Analysis (LCA), we examined the energy and CO_2 abatement (the second externality) potential of the three crops grown on the metal contaminated soils. We took into account the marginal impact of the metals in the biomass on the energy conversion efficiency and on the potential use of the rest products after conversion. Our analysis shows that digestion of energy maize with combustion of the contaminated digestate shows the best energetic and CO_2 abating perspectives. The replacement of cokes based electricity by willow is more efficient in CO_2 abatement than willow used in a CHP unit, despite lower net energy production in the former option. Willow reaches the same energy production and same CO_2 abatement per hectare per year as energy maize when its relative biomass yield (to energy maize) is respectively 0.64 and 0.44.

Further calculations on economic valuation indicated whether subsidizing the use of biomass harvested on contaminated soils would be economically efficient. Our case study indicates energy maize and rapeseed as the economically and energetically most valuable crops. Existing energy subsidies are already reflected in today’s crop prices. Therefore, CO_2 abatement potential should not be included again in the biomass price as this would lead to double counting. Our calculations, based on the “true” price per ton of biomass and the price per GJ to internalize the CO_2 benefit, suggest that current Flemish subsidies for renewable energy production are not generating the right price signals. Implications for the phytoremediation technology are mixed. We found that these true prices are not high enough to encourage renewable energy production, whether contaminated or uncontaminated biomass is used. However, the analysis clearly indicates that including CO_2 benefits would increase phytoremediation’s competitive advantage over conventional remediation technologies. Our findings support phytoremediation’s label of being sustainable, which could support its introduction on a commercial scale.

In **Section 4** the crop choice model was built around 3 alternative crops and 16 high income crops. Based on food- and fodder threshold values, determined as the intersection of marginal damage cost (MDC) of metals in crops on human health and the precautionary principle, the maximum Cd concentration in soil (C_q) for which these high income crops could be grown were determined. The outcome of the optimization model is the combination of one remediation crop and one HI crop for every initial level of contamination (C_0), resulting in an optimal end level of contamination (C_x), maximizing net benefit (R, € ha^{-1} over infinity, and at an interest rate of 4%).

Given the contamination range (12-0 mg kg^{-1}), the model only suggests growing one of the three crops in the context of risk reduction in 15% of cases. Main reasons for this are the fact that it is theoretically safe to grow maize for the grain (!) from 10 mg kg^{-1}, and the fact that, when C_0 already lies below C_q for a HI crop, the model often does not suggest growing one of the three alternative crops (instead, the model suggests immediately safely growing the allowed HI crop). Energy maize is the preferred alternative crop for large distances to target (DTT), *i.e.* the difference between C_q and C_0, while willow is preferred for average to small DTT. A raise in the biomass yield of willow results in willow to be chosen more often as the alternative crop compared to energy maize and rapeseed, but also results in willow to be suggested for higher DTT. To induce actual remediation in 50% of cases, the adapted gross income (AGI) of willow should at least be € 1,200 ha^{-1} year^{-1}. Our analyses show that this could not be reached by changing parameters separately, but a simultaneous increase in allowed rotation cycles, an increase in biomass yield, a reduction in costs, and the harvest of leaves might be able to realize this.

These results are coming from the model based on food- and fodder threshold values. Given $C_0 > 2$ mg kg^{-1}, which is the soil standard above which soil should be remediated (BSN), the model never suggests to remediate soil until $C_x < 2$ mg kg^{-1}, simply because this is not the economically optimal solution. Contamination is concentric around the smelters. It could therefore also be expected that farmers in the region are facing different concentrations of Cd, with farmers in the near vicinity of the smelters facing highest Cd concentrations. Resultingly, the use of soil standards will benefit some farmers, whereas the use of food- and fodder threshold values will benefit others. Indeed, the application of food- and fodder threshold values is beneficial for farmers when C_0 lies between 10.8 and 1.5 mg kg^{-1} soil. Comparing different soil standards, we notice that this only results in a shift in R over the initial concentrations (C_0): the exact same R is reached at a higher C_0, in case soil standards are less strict. Since soil standards are much lower than in the case of food- and threshold values, remediation will at any case take very long. This is a disadvantage for willow, as this crop benefits from average to short DTT. As a result, energy maize is most often preferred in case the optimization model is based on soil standards.

In choosing energy maize one opts for a long-term scenario. It is a choice for sustainable land use instead of remediation as such. Intuitively, energy maize would be preferred by farmers since it is a familiar crop requiring almost no adjustments. We suggest growing energy maize to aim at risk reduction, to generate an alternative income for farmers, and in the very long run also to obtain a gradual reduction of pollution levels. In this way, remediation is demoted to a secondary objective with sustainable risk based land use as a primary objective. We zoom in on this crop and find that the effect on the income per hectare of growing energy maize instead of fodder maize seems positive. The probability on a positive extra income is 90% when the farmer grows energy maize instead of fodder maize and simultaneously exploits the digester in cooperation with other farmers.

One of the drawbacks of the optimization model lies in its practical applicability. The model optimizes for each Cd concentration in soil (C_0) the income over infinity. The infinite time line does not appeal to current farmers. Moreover, a different approach for each of the hectares does not take into account the economies of scale, *i.e.* cultivation costs will be relatively higher for 1 hectare land than for 18 hectares of land with the same crop. Also, the model is so theoretic that it might be hard to understand what the actual implications for farmer might become. Taking into account the previous comments on the acceptability of crops, a second model ("farm based model") is developed that starts with a given number of hectares of willow combined with traditional and easily acceptable crops. This model calculates maximum DTT, and within these DTT, maximizes R per DTT (instead of per C_0) by changing this crop scheme, given the limited time period of 40 years. A first important conclusion from this second model (which is no longer an optimization model, but rather analyzes whether phytotechnologies can be used in a more acceptable approach with gradual integration of remediation crops in the crop scheme) is that phytoremediation is in any case more cost-effective than conventional remediation, within limited contamination ranges. A second important conclusion is that SRC of willow shows the physical and economic (if combined with HI crops) potential to sustainably remediate moderately Cd enriched land within a period of 40 years, with calculations even showing increases in R for the farmer at small DTT. A third conclusion from the second model is that remediation will at any case cost money, but with the correct incentive, government can guide the intensity of remediation.

SAMENVATTING

Wanneer, om financiële of technische redenen, conventionele bodemsaneringstechnieken niet mogelijk zijn, wordt in literatuur vaak fytoremediatie als alternatieve technologie naar voren geschoven. Bovendien zou het gebruik van de geoogste biomassa afkomstig van verontreinigde gronden als grondstof voor energieproductie of (chemische) industrie kunnen bijdragen tot een reductie in CO_2 emissies, niet enkel in vergelijking met conventionele saneringstechnologieën, maar eveneens in het kader van de Europese Richtlijn Hernieuwbare Energie. Echter, de geoogste biomassa kan verhoogde concentraties aan metalen bevatten en dient daarom met de nodige voorzichtigheid te worden gebruikt of zelfs verwijderd opdat de metalen zich niet opnieuw in het milieu zouden verspreiden en bijhorende gezondheidskosten met zich zouden meebrengen. De economische analyse van al deze aspecten verbonden aan fytoremediatie werd nog nooit (integraal) uitgevoerd.

Er bestaat geen twijfel dat fytoremediatie het nodige potentieel heeft om de degradatie en/of verwijdering van organische vervuiling uit grond en ondiep grondwater te bevorderen (Vangronsveld *et al.*, 2009; Weyens *et al.*, 2009, 2010). Echter, op basis van extrapolaties van data uit potexperimenten heeft men (te) enthousiaste beloften gemaakt met betrekking tot de mogelijkheden van metaalfytoextractie (Salt *et al*., 1995; Rulkens *et al.*, 1998; Susarla *et al*., 2002; Mench *et al.*, 2010). Algemeen bestaat een spoorelement fytoextractie protocol uit de volgende elementen (Vangronsveld *et al.*, 2009):

(i) teelt van de geschikte plant op de verontreinigde site;

(ii) verwijdering van de oogstbare plantendelen aangerijkt met de spoorelementen; en

(iii) nabehandeling van de geoogste plantendelen (zoals vergisting, persing, verbranding).

Met betrekking tot (i) is de sleutelvraag sinds de introductie van het fytoextractie van spoorelementen concept: "Kiest men voor planten die de beoogde spoorelementen hyperaccumuleren of voor planten met een hoge biomassa productie?" Met betrekking tot (ii) en (iii) zijn de grootste hinderpalen voor de commerciële doorbraak van fytoextractie van spoorelementen de lange saneringsduur om de concentraties in de bodem naar wettelijk aanvaarde standaarden te reduceren, en het gebruik, de valorisatie, en/of verwijdering van de verontreinigde biomassa.

Elke verontreiniging is verschillend en succesvol duurzaam beheer vereist een zorgvuldige integratie van alle relevante factoren. Er is nood aan meer data om het economische plaatje te kwantificeren, en aan meer demonstratieprojecten rond metaal fytoextractie om aanbevelingen te kunnen maken en het beleid en de publieke opinie te overtuigen van de mogelijkheid van fytotechnologieën als alternatief voor conventionele saneringstechnieken voor de behandeling van gronden, brownfields, grondwater en afvalwater vervuild met toxische metalen (en organische polluenten) (Ruttens en Vangronsveld, 2006; Adriaensen *et al.,* 2008; Vangronsveld *et al.*, 2009). Het is daarom noodzakelijk dat een duidelijk model wordt uitgewerkt voor het gebruik van fytotechnologieën op basis waarvan de gebruiker tot een gefundeerde beslissing kan komen om al dan niet fytotechnologieën toe te passen, met het meest geschikte gewas gevolgd door de meest geschikte bewerking (Adriaensen *et al.*, 2008). We ontwikkelden daarom een optimalisatie instrument voor de evaluatie van verscheidene gewas – verwerkingscombinaties, gebaseerd op een matig metaal verontreinigde landbouwgrond in de Kempen (700 km² in België en Nederland). Het strategische doel voor deze regio bestaat uit het herstel van voornamelijk met cadmium (Cd) aangerijkte landbouwgronden met behulp van planten die niet enkel saneren, maar tevens een inkomen voor de landbouwer genereren.

In **Sectie 1** bespreken we de achtergrondsituatie in de Kempen, alsook de huidige federale en regionale wetgeving rond bodemsanering, en de economische evaluatie van overheidsinterventie ter bescherming van het milieu en de algemene gezondheid. Onze analyses beperken zich tot 3 (energie)gewassen: energiemaïs (*Zea mays*), koolzaad (*Brassica napus*), en korte omloop wilg (*Salix spp.*) op basis van de reeds bestaande ervaring met deze planten en de beschikbaarheid van data afkomstig van het experimentele veld in de Kempen.

Verhoogde metaalconcentraties in de grond zorgen voor verhoogde metaalconcentraties in de plant. Een cruciaal aspect bij gewasteelt op verontreinigde grond is de vraag of de geoogste biomassa zal beschouwd worden als (gevaarlijk) afval of biomassa, vermits deze classificatie een effect heeft op verder gebruik en valorisatie van het gewas. Daarom, alvorens een fytoremediatie beslissingsmodel kan worden ontwikkeld, ontwarren we in **Sectie 3** de vele definities en reguleringen gerelateerd aan het concept en het gebruik van met metalen aangerijkte energiegewassen, vermits dit technische en dus economische gevolgen kan hebben voor verwerkingsprocessen. Wij vonden in bestaande wetgeving geen reden om energiegewassen afkomstig van metaal verontreinigde grond niet als biomassa te beschouwen. Wij stellen voor dat de classificatie van energiegewassen met metaal accumulerende doeleinden afhangt van de hoofddoelstelling van de teelt. Indien deze duurzaam gebruik van landbouwgrond met behulp van energiegewassen betracht, dan zouden de gewassen beschouwd worden als biomassa. Fytoremediatie is enkel de hoofddoelstelling bij gewassen die metalen accumuleren met substantiële dalingen in metaalconcentraties in de bodem tot gevolg (zoals bij hyperaccumulatoren). Gegeven de relatief lage accumulatiecapaciteit van de bestudeerde planten, veronderstellen we het duurzaam gebruik van verontreinigde landbouwgrond met inkomensgeneratie op basis van energieproductie als hoofddoelstelling. Voor hout blijkt het onderscheid naar afval of biomassa van belang te zijn voor de toepasselijke wetgeving met betrekking tot het type verbrandingsinstallatie en emissiecontrole. Voor energiemaïs en koolzaad vonden we geen expliciete wetgeving met betrekking tot de input voor energieconversie, maar er zijn richtwaarden voor het restproduct na conversie.

Voor de drie planten in onze gevalsstudie vonden we zelfs geen marginaal effect van de metalen op de energieconversie (technische) efficiëntie. Bovendien eindigen de metalen niet in de energiedrager, maar worden ze geconcentreerd in het restproduct: assen, digestaat en cake. Onveilig gebruik en verwijdering zouden kunnen leiden tot een substantieel verhoogd risico voor mens en milieu. De aanwezigheid van toxische metalen in het residu na conversie van de biomassa zijn een schoolvoorbeeld van een externaliteit omdat hun aanwezigheid niet gereflecteerd wordt in de prijs van de biomassa. We bestuderen de impact van de verontreiniging op de bewerking van de restproducten voor secundair gebruik of de verwijdering ervan omdat men, om aan de huidige wetgeving te voldoen, eventueel op zoek zal moeten gaan naar een alternatief eindgebruik voor (een deel van) het restproduct. Bij strikte toepassing van de huidige wetgeving leidt toepassing van een klein deel van het digestaat op het land en verbranding van het overgrote deel tot de hoogste economische waarde. Voor koolzaadkoek hebben de metalen geen impact op het gebruik als voeder. Wilgenassen dienen te worden gestort omwille van te hoge metaalconcentraties. Het verschil in economische waarde tussen het gecontamineerde en niet gecontamineerde restproduct wordt meegenomen naar het optimalisatie model.

Met behulp van een levenscyclusanalyse (LCA) onderzochten we het potentieel van de drie gewassen, geteeld op metaal verontreinigde grond, om hernieuwbare energie te genereren en om de CO_2 uitstoot (externaliteit) te reduceren. We hielden rekening met de marginale impact van de metalen op het energieconversieproces alsook met het potentieel gebruik van de restproducten na conversie. Vergisting van energiemaïs gecombineerd met de (hoofdzakelijke) verbranding levert het meeste netto energie op en vermijdt bovendien het meeste CO_2 uitstoot. De vervanging van cokes door wilg voor elektriciteitsproductie resulteert in een hogere reductie in CO_2 uitstoot dan het gebruik van wilg in een warmtekrachtkoppeling installatie voor de gelijktijdige opwekking van warmte en elektriciteit, ondanks een lagere netto energie productie in de eerstgenoemde optie. Een bijkomende vaststelling is dat wilg dezelfde energieproductie en vermindering in CO_2 uitstoot zou kunnen teweegbrengen als energiemaïs indien zijn biomassaverhouding ten opzichte van dit gewas respectievelijk 0.64 en 0.44 zou zijn.

Vervolgens gingen we na of de subsidiëring van de biomassa op basis van de economische valorisatie van deze verminderde CO_2 uitstoot economisch efficiënt is. Bestaande energiegerelateerde subsidies worden vandaag reeds gereflecteerd in gewasprijzen. Daarom dient het feit dat de gewassen CO_2 uitstoot kunnen reduceren niet meer opgenomen worden in de biomassa prijs, vermits dit zou leiden tot een dubbeltelling. Onze berekeningen gebaseerd op de werkelijke waarde per ton biomassa en per GJ suggereren echter dat Vlaamse subsidies voor hernieuwbare energieproductie niet de juiste prijssignalen zenden. Gevolgen voor fytoremediatie zijn gemengd. We stelden vast dat de subsidies gebaseerd op werkelijke prijzen enerzijds niet hoog genoeg zijn om hernieuwbare energieproductie aan te moedigen, ongeacht of niet gecontamineerde of gecontamineerde biomassa wordt gebruikt. Echter, de analyse geeft duidelijk aan dat het opnemen van CO_2 reductie het competitief voordeel van fytoremediatie ten opzichte van conventionele technieken zou verhogen. Onze bevindingen ondersteunen bovendien het duurzame karakter van fytoremediatie, wat zijn implementatie op commerciële schaal zou kunnen ondersteunen.

In **Sectie 4** wordt het optimalisatiemodel opgebouwd rond de drie gewassen en 16 hoge inkomens (HI) gewassen. Gebaseerd op voedsel- en voedernormen, bepaald als de kruising tussen marginale schade kosten van metalen in HI gewassen op de menselijke gezondheid, en het voorzichtigheidsprincipe, werd de maximale Cd concentratie in de bodem (C_q) bepaald waarbij deze HI gewassen mogen worden geteeld. De uitkomst van het optimalisatie model is voor elke initiële Cd concentratie in de grond (C_0) een combinatie van 1 van de drie gewassen gevolgd door een HI gewas bij een optimale eind contaminatie (C_x) die het netto resultaat maximaliseren (R, € ha^{-1} over een oneindige tijdsduur en aan een intrestvoet van 4%).

Gegeven het verontreinigingsspectrum waarbinnen het model is opgesteld (12-0 mg kg^{-1}) stelt het model slechts één van de drie alternatieve gewassen voor in de context van risicoreductie in 15% van de gevallen. Hoofdredenen hiervoor zijn het feit dat het theoretisch veilig is om korrelmaïs te zetten vanaf 10 mg kg^{-1} en het feit dat wanneer C_0 reeds onder een C_q ligt het model vaak niet voorstelt om éé van de drie alternatieve gewassen te zetten, maar om onmiddellijk het toegelaten HI gewas bij deze C_q. Energiemaïs wordt verkozen indien grote afstanden in Cd concentraties (DTT) (tussen C_0 en C_q) dienen overbrugd te worden alvorens veilig een HI gewas kan worden gezet, terwijl wilg het uitgelezen gewas is bij middelmatige tot kleine DTT. Verhoging van de biomassaopbrengst van wilg zorgt ervoor dat wilg vaker wordt verkozen als remediërend gewas en dat wilg wordt ingezet bij hogere DTT. Opdat in 50% van de gevallen zou overgegaan worden tot werkelijke sanering, zou het aangepast bruto inkomen (AGI) van wilg ten minste € 1,200 € ha^{-1} $jaar^{-1}$ moeten bedragen. Onze analyses tonen aan dat dit niet mogelijk is door afzonderlijke wijziging van parameters, maar wel door een gelijktijdige stijging in rotatiecycli, een verhoging in biomassa opbrengst, een reductie in kosten, en oogst van de bladeren.

Deze resultaten zijn afkomstig wanneer het model gebaseerd is op voedsel- en voedernormen. Indien $C_0 > 2$ mg kg^{-1}, de bodemnorm boven dewelke de bodem zou moeten geremedieerd worden (BSN), dan stelt dit oorspronkelijke model nooit voor om te remediëren tot $C_x < 2$ mg kg^{-1}, omdat dit niet de economisch meest optimale oplossing is. De verontreiniging is concentrisch verspreid rond de fabrieken. Men mag dan ook verwachten dat landbouwers in de regio te maken hebben met verschillende Cd concentraties, hoe dichter bij de bron, hoe hoger de Cd concentratie. Bijgevolg zal het gebruik van bodemnormen voordelig zijn voor sommigen, terwijl voor anderen voedsel- en voedernormen voordeliger zijn. Inderdaad, voedsel- en voedernormen zijn voordelig voor landbouwers met een initiële verontreiniging (C_0) tussen 10.8 en 1.5 mg Cd kg^{-1}. Wanneer we bodemstandaarden onderling vergelijken stellen we vast dat dit enkel leidt tot een shift in R voor de verschillende C_0. Aangezien bodemnormen ver beneden voedsel- en voedernormen liggen, zal sanering in ieder geval lang duren. Dit is een nadeel voor wilg, vermits dit gewas profiteert van kleine tot middelmatige DTT. Bijgevolg is energiemaïs het meest gekozen gewas in het model gebaseerd op bodemnormen.

Door te kiezen voor energiemaïs kiest men voor een lange termijn scenario. Het is de keuze voor duurzaam bodemgebruik eerder dan voor sanering. Intuïtief wordt energiemaïs verkozen door landbouwers vermits zij vertrouwd zijn met het gewas en het bijna geen aanpassingen op het landbouwbedrijf met zich meebrengt. Energiemaïs kan echter enkel worden geteeld met het oog op risicoreductie, om een alternatief inkomen voor de landbouwers te garanderen, en om op (zeer) lange termijn een stapsgewijze daling van de vervuiling te bekomen. Meer gerichte analyses tonen aan dat het effect op het landbouwinkomen per hectare door de vervanging van voedermaïs door energiemaïs positief is. De kans op een positief extra inkomen door deze omschakeling, gecombineerd met de uitbating van een vergister in samenwerking met andere landbouwers is bovendien 90%.

Eén van de nadelen van het eerste optimalisatiemodel ligt in zijn praktische toepasbaarheid. Het model optimaliseert voor elke C_0 het inkomen over een oneindige tijdsperiode. Deze oneindige tijdsperiode spreekt landbouwers niet aan. De gedifferentieerde aanpak per hectare houdt evenmin rekening met schaalvoordelen, teeltkosten zullen namelijk relatief hoger liggen voor 1 hectare land dan voor 18 hectares land met hetzelfde gewas. Bovendien is het model te theoretisch om eenvoudig verstaanbaar te zijn. Rekening houdend met deze vaststellingen en met de aanvaardbaarheid van bepaalde gewassen werd een tweede model ontwikkeld dat vertrekt vanuit een gegeven aantal hectares wilg gecombineerd met meer traditionele en aanvaardbare gewassen. Dit tweede model berekent eerst de maximale DTT die binnen 40 jaar kan overbrugd worden met wilg en zoekt vervolgens naar het optimale gewasschema dat leidt tot een maximale R (dus per DTT en niet meer per C_0). Een eerste belangrijke vaststelling uit dit tweede model (dat niet langer een optimalisatiemodel is, maar eerder nagaat of fytotechnologieën kunnen toegepast worden in een meer aanvaardbare benadering met graduele integratie van remediërende gewassen in het gewasschema) is dat fytoremediatie in elk geval meer kosten efficiënt is dan conventionele sanering, gegeven beperkte DTT's. Een tweede vaststelling is dat korte omloop wilg het fysieke en economische potentieel heeft (indien gecombineerd met HI gewassen) om op een duurzame manier matig Cd verontreinigd land te saneren binnen een tijdsbestek van 40 jaar, met berekeningen die zelfs een stijging in R aantonen voor de landbouwer voor kleine DTT. Een derde vaststelling in het tweede model is dat sanering in elk geval geld zal kosten, maar met de juiste aanmoediging kan de overheid de intensiteit van de sanering sturen.

TABLE OF CONTENTS

ACKNOWLEDGEMENTS i
SUMMARY v
SAMENVATTING xiii
TABLE OF CONTENTS xxiii
LIST OF ABBREVIATIONS xxx

Section 1 39
Chapter 1.1 History of soil contamination in the Campine region 43
Chapter 1.2 Phytoremediation 47
1.2.1 Theory 47
1.2.2 Practice 49
Chapter 1.3 Economic decision theory 55
1.3.1 Cost-Benefit Theory and Cost-benefit analysis 55
1.3.1.1 General 55
1.3.1.2 The pursuit of efficiency 56
1.3.1.3 Underlying assumptions of CBT 57
1.3.2 Market failures/Missing markets with environmental consequences 58
1.3.2.1 Public goods 60
1.3.2.2 Externalities 61
1.3.3 Valuation techniques 64
1.3.4 Alternatives to CBA 67
1.3.5 Does CBA guarantee sustainability 71
1.3.5.1 Sustainability 71
1.3.5.2 Discounting and the choice of a social discount rate 73
1.3.6 Solutions to externalities 76
1.3.6.1 Property rights as a potential solution 76
1.3.6.2 Command and control (setting standards) 78
1.3.6.3 Performance oriented approach 80
1.3.6.4 Economic based incentives 80
1.3.6.5 Uncertainty regarding MAC and MDC curves 85
1.3.6.6 Should victims be compensated? 88
1.3.6.7 Critical analysis of externality theory 90

1.3.7 Conclusion 92
Chapter 1.4 Legislative issues related to the Campine case 95
1.4.1 European Soil Strategy and the new Soil Framework Directive 95
1.4.2 Soil remediation threshold values 96
1.4.3 European and Belgian product threshold/safety values 101
Section 2 107
Introduction 109
Multiple layer problem 110
Research questions 113
Decision model 116
Challenges 118
Phytoremediation 119
Correcting for externalities 120
Soil quality based on legislation 123
Section 3 129
Chapter 3.1 Private costs and benefits of plant-based technologies 131
3.1.1 Experimental site 131
3.1.1.1 Background 131
3.1.1.2 Initial Cd concentration in soil (C_0) 134
3.1.1.3 Concentrations and biomass yield in alternative crops 135
3.1.1.4 Energy options for biomass from the experimental site 139
3.1.2 Economic valuation of alternative crops and HI crops 139
3.1.2.1 Common Agricultural Policy (CAP) 139
3.1.2.2 Adapted gross income 141
3.1.2.3 Current agricultural activities 144
3.1.2.4 Transport 146
3.1.2.5 Rapeseed 147
3.1.2.6 Energy maize 149
3.1.2.7 SRC of willow 152
3.1.2.8 Overview 154
3.1.3 Economic valuation of high income crops 155
3.1.3.1 AGI_{HI} 155
3.1.3.2 Rotation schemes 157

3.1.3.3 AGI'$_{HI}$ ... 159
Chapter 3.2 Metal enriched biomass for energy ... 161
Abstract ... 161
3.2.1 Introduction ... 163
3.2.2 Classification of biomass from plant-based technologies and used for energy production ... 164
3.2.2.1 Energy maize ... 168
3.2.2.2 Rapeseed ... 169
3.2.2.3 Willow ... 170
3.2.2.4 Conclusion on input ... 173
3.2.3 Energy conversion of biomass grown on metal enriched soils in the Campine ... 173
3.2.3.1 Policy on emissions of metals ... 173
3.2.3.2 Effect of metals on energy production potential ... 176
3.2.3.3 Conclusion on energy conversion ... 180
3.2.4 Marginal impact of metals on end use of rest product ... 181
3.2.4.1 Waste Policy ... 181
3.2.4.2 Secondary use of digestate ... 185
3.2.4.3 Secondary use of rapeseed cake ... 188
3.2.4.4 Secondary use of ashes ... 189
3.2.4.5 Conclusion on legislation rest products ... 193
Chapter 3.3 Economic analysis of rest products ... 195
Abstract ... 195
3.3.1 Introduction ... 196
3.3.2 Data and methods ... 197
3.3.2.1 Metals in biomass as an externality of plant-based technologies ... 197
3.3.2.2 General approach ... 198
3.3.2.3 Assumptions in the economic analysis ... 201
3.3.3 Results ... 202
3.3.3.1 Economic valuation of digestate ... 202
3.3.3.2 Economic valuation of rapeseed cake ... 210
3.3.3.3 Economic valuation of willow ashes ... 213
3.3.3.4 Destination of ashes after combustion of rest products ... 216
3.3.3.5 Transport of rest products, glycerin and ashes ... 217

3.3.4 Discussion and conclusion ... 219
Chapter 3.4 Energy production and carbon dioxide abatement of crops for sustainable soil management ... 223
Abstract ... 223
3.4.1 Introduction ... 224
3.4.2 Data and methods ... 229
3.4.2.1 Assumptions ... 231
3.4.2.2 Fossil energy input for biomass production ... 232
3.4.2.3 From biomass to gross energy ... 233
3.4.2.4 Effect of metals on energy production potential ... 236
3.4.2.5 From gross energy content to net thermal, electric, and mechanical energy 237
3.4.2.6 Metal enriched rest product ... 238
3.4.2.7 Overview ... 241
3.4.3 Results ... 243
3.4.3.1 Net energy ... 243
3.4.3.2 CO_2 abatement ... 246
3.4.4 Discussion and conclusion ... 247
Chapter 3.5 Economic assessment and policy analysis of CO_2 abatement . 253
Abstract ... 253
3.5.1 Introduction ... 254
3.5.2 Data and methods ... 258
3.5.2.1 Economic valuation: government intervention ... 258
3.5.2.2 Shedding light on the shadow price of carbon ... 260
3.5.2.3 Avoid double counting of external costs of energy production ... 263
3.5.3 Results ... 266
3.5.3.1 Private economic results ... 266
3.5.3.2 CO_2 abatement ... 267
3.5.4 Discussion and conclusion ... 272
Section 4 ... 279
Chapter 4.1 Crop choice model ... 281
Abstract ... 281
4.1.1 Introduction ... 283
4.1.2 Data and methods ... 285

4.1.2.1 Initial level of soil contamination (C_0) 285
4.1.2.2 Final level of soil contamination 285
4.1.2.3 Remediation duration 288
4.1.2.4 Overview of economics of alternative crops 290
4.1.3 Model 291
4.1.3.1 General overview 291
4.1.3.2 Yearly accumulation per crop per hectare: $BP_i \cdot E_i$ 294
4.1.3.3 Total accumulation by crop i: $REM_i = Q_0 - Q_q = A \cdot d \cdot \rho \cdot (C_0 - C_q) = A \cdot t_i \cdot BP_i \cdot E_i$ (Eq. 19) and (Eq. 20) 295
4.1.3.4 Remediation duration: $t_i = (Q_0 - Q_q)/(BP_i \cdot E_i)$ 295
4.1.3.5 AGI' with corrections for externalities based on current legislation . 298
4.1.3.6 NPV(r) = R 299
4.1.3.7 Max $R_{rem} + R_{HI}$ 299
4.1.4 Results 303
4.1.4.1 Base case 303
4.1.4.2 Alternative crops 304
4.1.4.3 pH 316
4.1.4.4 Non linearity of metal accumulation 319
4.1.4.5 Extraction potential 322
4.1.5 Discussion and conclusion 323
4.1.5.1 Model outcomes based on food- and fodder threshold values 323
4.1.5.2 Legislation on soil remediation: inconsistent? 324
4.1.5.3 Legislation on soil remediation: inefficient? 328
4.1.5.4 Alternative to the precautionary principle 333
4.1.5.5 Comparison of soil standards and food threshold values 336
4.1.5.6 Uncertainty level 340
4.1.5.7 Compensation of farmers 342
Chapter 4.2 Energy maize as an acceptable alternative crop for risk management of contaminated land 353
Abstract 353
4.2.1 Introduction 354
4.2.2 Economics of switching from fodder to energy maize 355
4.2.2.1 Data and methods 355
4.2.2.2 Results 358

4.2.3 Economics of anaerobic digestion of energy maize 364
4.2.3.1 Data and methods 364
4.2.3.2 Results 368
4.2.4 Conclusion 378
Chapter 4.3 Farm based model 387
Abstract 387
4.3.1 Introduction 387
4.3.2 Data and methods 389
4.3.2.1 Physical assumptions 389
4.3.2.2 Economic assumptions 393
4.3.3 Results 394
4.3.3.1 Base case 394
4.3.3.2 Sensitivity analysis 398
4.3.4 Discussion and conclusion 403
Section 5 411
Chapter 5.1 Introduction 413
Chapter 5.2 Research questions answered 415
5.2.1 Does phytoremediation result in other than private costs and benefits and if so, are these externalities positive or negative? 416
5.2.1.1 CO_2 abatement potential of biomass for energy 416
5.2.1.2 Metal contaminated biomass 418
5.2.2 Do current policies internalize in an economically efficient way the externalities of soil pollution? Do current policies correctly internalize the externalities of phytoremediation? 422
5.2.2.1 Food- and soil policy 422
5.2.2.2 Energy policy 426
5.2.2.3 Metal waste disposal policy 428
5.2.3 Does phytoremediation offer an economically viable alternative for conventional remediation technologies and which crops should be used and under what circumstances? 428
5.2.4 Does phytoremediation offer a multifunctional and sustainable alternative for conventional remediation technologies for functional repair or management of metal contaminated agricultural sandy soil? 433
Chapter 5.3 Future research 434
LITERATURE 441
List of publications 477

LIST OF ABBREVIATIONS

A	area surface
AF	annuity factor
AGI, AGI’	adapted gross income, AGI corrected for rotation scheme
ALV	agency for agriculture and fisheries
As	arsenic
BAT(NEEC)	best available technology (not entailing excessive cost)
BCF	bio concentration factor
BP_i	biomass production per hectare per year of crop i
BSN	soil remediation standard
BV	background value
C	equipment costs
C_0	initial Cd concentration in soil
C_{ix}	concentration in soil in year x, using crop i
C_q	concentration Cd (mg kg^{-1} soil) to comply with food norms
C_{qs}	concentration Cd (mg kg^{-1} soil) equal to BSN
CAL_W	calorific value of willow
CAP	common agricultural policy
Caracas	concerted action on risk assessment for contaminated sites
CBA / CBT / CEA	cost-benefit analysis/ cost-benefit theory/ cost-effectiveness analysis
Cd	cadmium
CEN	European committee for standardization
CF	cash flow
CH_4	methane
CHP	combined heat and power
Clarinet	contaminated land rehabilitation network
CLO	collective agricultural research
CO_2	carbon dioxide
COD	chemical oxygen demand
COM	combustion
Cr	chromium
Cs	caesium

Cu	copper
d	soil depth
D	third Party Labor costs
D_i	disposal cost of product i (€ ha^{-1} y^{-1})
D_{BD}	density of biodiesel
D_{PPO}	density of PPO
DALY	disability adjusted life years
DIG	digestion
Dis	distance
dm	dry matter
dm%	dry matter percentage
DPSI-R	driving forces-pressures-state-impacts-responses
DTT	distance to target
e_u	initial level of external effect (pollution/emission)
e*	final level of external effect (pollution/emission)
E	herbicides, fertilizer and planting material costs
E_{ix}	heavy metal concentration in crop i in year x
E_{post}	($E_{post\ out\ i\ C}$) minus ($E_{post\ in\ i\ C}$)
$E_{post\ in\ i\ C}$	energy input for end use i of contaminated rest product
$E_{post\ out\ i\ UC}$	energy output for end use i of uncontaminated rest product
EB_{EN}, EB_M	external benefit of CO_2 abatement and metal waste disposal
EC_{EN}, EB_M	external cost of CO_2 abatement and metal waste disposal
ECF	European common forum
ECN	energy research centre of the Netherlands
EI	total primary energy input
EI_{pred}	direct energy input during crop establishment, incl. transport
EI_{prei}	indirect energy input during establishment of the crop
EM	energy maize
EN	renewable energy
EO	total energy output
EPA	environmental protection agency (United States)
EPER	European pollutant emission register
ER	energy ratio (EO/EI)
ETS	emissions trading scheme

EU	European Union
EUA	European Union allowances
EV_{BD}	energy value of biodiesel
EV_{BG}	energy value of biogas
EV_{PPO}	energy value of PPO
F	crop and animal related costs
FAME	fatty acid methyl esters
FAO	food and agriculture organization
FAVV	federal agency for food safety
fm	fresh matter
FO	fodder
FOD	federal public service
g	1 (incl. leaves); 0 (no leaves)
G	fuel costs
G_{BD}	efficiency of rapeseed conversion to biodiesel
G_{EM}	biogas yield of energy maize
G_{PPO}	efficiency of mechanical rapeseed pressing to PPO
GCC	green (electricity) certificate
GEC_i	gross energy content of crop i after conversion
GER	gross energy requirement (EO+EI)/EI
GHG	greenhouse gas
GIS	geographic information system
GJ	gigajoule (10^3 MJ)
GRAN	granulates
GWP	global warming potential
h	hour
H	total revenue per hectare per year
ha	hectare
Hg	mercury
HI (crop)	high income (crop)
HM	heavy metal
i	discount rate
IEABCC	international energy agency for biomass combustion and co-firing

INBO	research institute for nature and forest
IPCC	intergovernmental panel on climate change
IPPC	integrated pollution prevention and control
IRR	internal rate of return
ISO	international standards organization
ITRC	interstate technology and regulatory council
IWT	institute for the promotion of innovation by science and technology
k	rotation cycle
km	kilometer
kWh	kilowatt hour
K.B.	royal decision
l	liter
LCA	life cycle analysis
LCP	large combustion plants
LF 1	landfill category 1
LF 2	landfill category 2
LHV	lower heating value
m%	relative contribution of various plant parts to total biomass
MAC	marginal abatement cost
MAP	manure action plan
MCDA	multi criteria decision analysis
MD	manure decree
MDC	marginal damage cost
MINA	Flemish environmental policy plan
MIP	environmental innovation platform
MIRA	Flemish environmental report
MJ	megajoule
Mn	manganese
MWe	megawatt electric
MWh	megawatt hour
N	nitrate
NE	net energy
Ni	nickel

Nicole	network for industrially contaminated land in Europe
Nm^3	normal cubic meters
NPV	net present value
OECD	organization for economic cooperation and development
OVAM	public waste agency Flanders
OVB	association for entrepreneurs in Belgium
OWS	organic waste systems
P	price
PAH	poly aromatic hydrocarbon
Pb	lead
PM	particulate matter
POVLT	provincial agri- and horticultural research and education centre
PPO	pure plant oil
q	successive years crop can be grown (asparagus:10, other:1)
Q	quantity
Q_0	metals initially present in the soil
Q_q	amount of metals in soil for which it is allowed to grow HI crops
Q_x	amount of metals in soil that is economically most viable
r	AGI adjusted for rotation scheme and external costs and benefits
R	r discounted at 4%
R_{HI} , R_{rem}	R after C_x , R before C_x
RE	renewable energy
REM_i	total amount of metals to be removed per hectare by crop i
RS	rapeseed
s(e)	subsidy at emission/pollution level e
S_{rem}	ton soil (for which DTT is covered)
SCAPE	soil conservation and protection in Europe
SCC	social cost of carbon
SCF	scientific committee on food
SME	small and medium enterprises
SO	standard output

SPC	shadow price carbon
SRC	short rotation coppice
SRTP	social rate of time preference
t_i	number of years of growth for crop i
t(e)	tax in emission/pollution level e
T	transport cost per ton per year
T_d	transport cost per ton per km per year
T_i	transport cost of product i per hectare per year
TDI / TWI	tolerable daily intake / tolerable weekly intake
TEV	total economic value
Tl	thallium
TV	target value
UC/C	uncontaminated/contaminated
USDA	United States department of agriculture
$V_{post\ i\ C}$	economic value of contaminated rest product after end use i based on CO_2 abatement $V_{EN\ post\ i\ C}$ plus market price substitute $V_{M\ post\ i\ C}$
V_t	social welfare at time t
VITO	Flemish institute for technological research
VLACO	Flemish compost organization
VLAREA	Flemish regulation concerning waste prevention and –management
VLAREBO	Flemish regulation on soil remediation and soil protection
VLAREM	Flemish regulation concerning the environmental permit
VMM	Flemish environment agency
VREG	Flemish Regulator of the Electricity and Gas market
W	willow
WF	waste framework
WHO	world health organization
WI	waste incineration
WTA/WTP	willingness to accept/willingness to pay
y	year
Z	change in income
Zn	zinc

α	loss of biomass/energy due to drying in open air/conditioned drying
$\beta(th)$, $\beta(el)$, $\beta(f)$	fossil energy use during conversion (resp. thermal, electric, diesel)
ξ	% fall in additional utility from each % increase in consumption
$\eta(th)$, $\eta(el)$	conversion efficiency (resp. thermal and electric)
Δ	marginal change
ρ	soil density

Section 1

INTRODUCTION

"A multidisciplinary approach is warranted to make phytoextraction a feasible commercial technology to remediate metal-polluted soils (do Nascimento and Xing, 2006)"

Phytoremediation is often mentioned as an economically viable, effective and environmentally sustainable remediation strategy (Kumar *et al.,* 1995; Salt *et al*., 1995; Rulkens *et al.*, 1998; Susarla *et al*., 2002; Liang *et al.,* 2009). The analysis should use a holistic approach, and take into account all potential opportunities that might evolve from the remediation of metal enriched agricultural land. Bardos *et al.* (2008) use the term "self-funding land management regime".

Parts of this section have been published in:

Witters, N., Van Slycken, S., Ruttens, A., Adriaensen, K., Meers, E., Meiresonne, L., Tack, F.M.G., Thewys, T., Laes, E., Vangronsveld, J. (2009) Short Rotation Coppice for phytoremediation of a Cd-contaminated agricultural area: A sustainability assessment. BioEnergy Research 2(3), p. 144-152

Chapter 1.1 *History of soil contamination in the Campine region*

From the end of the 19th century until mid 1970s, Zinc (Zn) and Lead (Pb) were refined using a pyrometallurgical process in the vicinity of Antwerp and in the Campine region in Belgium (Vangronsveld *et al*., 1995; Hogervorst *et al.*, 2007). This refining was done on sites in Balen/Lommel, Overpelt, Hoboken, and Olen. Metal fumes were condensed in a condenser, collected and transmitted to moulds. Volatile metals that were not collected, condensed on dust particles and were emitted to the air via the smokestack. As a result, a large area of more than 700 km² in Belgium and the Netherlands is moderately contaminated due to atmospheric deposition of the dust. Cadmium (Cd), Pb and Zn are the main pollutants (Hogervorst *et al*., 2007). These metals are found in the upper layer of the soil (30-40 cm) over an extended area, which in the Belgian part alone covers 280 km² (=28,000 ha) (www.ovam.be) (Table 1-1).

Table 1-1 Metal enriched areas in the Belgian Campine

Contamination level	*Area*	*Surface*
>1 mg kg^{-1}	Lommel, Overpelt, Neerpelt, Hamont-Achel	280 km²
>3 mg kg^{-1} †	Union Minière (Balen), Zn smelter (Lommel)	52 km²
	Union Minière (Overpelt)	16 km²

†within this zone, there are heavily contaminated zones of more than 6 and 12 mg Cd per kg soil

Moreover, the residues (ashes, slags) of this process were used for the hardening of roads, industrial zones, and private properties in residential areas. In Belgium and the Netherlands respectively at least 490 and 833 km of road were constructed with Zn, Cd and Pb containing residues, thereby spreading these metals far away from the originating smelters. Lastly, the dumping of effluent of the smelting activities in the surface water has heavily contaminated several water soils. Sludge has been deposited at the riversides (due to flood or deepening of the river bedding) and thereby the metals are widely spread. Percolation of the metals has also contaminated deeper soil layers and the groundwater (www.ovam.be).

The high exposure area (*i.e.* the area with an expected soil concentration of Cd of 3 mg kg^{-1}) borders on three Zn smelters, and consists of the municipalities Mol, Balen, Lommel, Overpelt, and Neerpelt.

As a result, these areas (urban and agricultural) were facing the externalities of the Zn production through atmospheric deposition of dust. Years later, government intervened, setting emission standards so that smelters would take into account Cd emissions in their marginal cost curve. However, when emission standards are set, there is no reason to believe that authorities will assign the responsibility for emission reduction in a cost-minimizing way (Tietenberg, 2003). This was indeed the case in Europe and Flanders. Emissions and immissions have been reduced, but not in a cost-effective way.

Cd emissions and immissions (in particulate matter, PM) are regulated by VLAREM II, respectively in Appendix 4.4.2 and 2.5.8. General emission threshold values for Cd have largely been taken over from TA-luft (Technical Instructions on Air Quality Control, Germany) and adapted through Best Available Techniques (BAT) analyses. VLAREM II then describes the actions which should be taken when a sector or company does not respect these standards (Claeys, N., personal communication, December 2009). Immission threshold values, *i.e.* values for air quality (5 ng m^{-3} to be realized in 2012) originate from a European Directive (2004/107/EC) relating to Cd in ambient air (Rosier, A., personal communication, October 2009). This Directive is based on a health impact study (EC, 2001) and on World Health Organization (WHO) Directives. If a Member State fails to live up to these values, the European Commission takes action concerning condemnation (Claeys, N., personal communication, December 2009).

While both standards guarantee that external costs (*e.g.* on health) of Cd emissions and immissions are internalized, they do not necessarily guarantee that this leads to a cost-effective and –efficient outcome. Standards are based on the precautionary principle, providing cautionary safety margins, and they are based on general standards and not on an agent by agent specific, economically based incentive approach. Moreover, there is no economic incentive for agents to live up to the values since the punishment is rather general (condemnation) and not economic as far as we know (EC, 2001).

We take a closer look at how the (Cd) emission policy in Europe, Flanders, and the Campine region proceeded. In 1992, the Rio Declaration gave impulse to the idea of setting up emission inventories as a tool for providing information on pollutants to the public. The idea was given concrete form in 1996 through the EU Directive on Integrated Pollution Prevention and Control (IPPC). In 2000, the details of the inventory were laid down in a Commission Decision (17 July 2000) on the establishment of a European Pollutant Emission Register (EPER). The first data set (2001, published 2004) and the second data set (2004, published 2006) contain data from 10,000 large and medium-sized industrial EU facilities, on their 50 main pollutant emissions to air and water, which exceed specified EPER emission threshold values[1]. To limit the administrative burden on industry, the EPER threshold values have been fixed to a level so as to cover about 90% of emissions from facilities within every IPPC region/country (www.eper.ec.europa.eu). The facilities with an obligation to report are mentioned in the Annex of the IPCC Directive. The installations in Overpelt and Balen fall under this Annex (metal processing industries).

Total yearly Cd emissions of both smelters in Overpelt and Balen reduced strongly in the last decade. In 1998, the emissions of Cd were respectively 97 and 8 kg year^{-1} in Overpelt and Balen. In 2004, this was only 14 kg year^{-1} for both smelters together (Peeters, 2006). Umicore Overpelt was urged to design a remediation plan in 1996. In 1998, filters were introduced on the emission

[1]Values are based on the size of the company. A company can satisfy emission and immission standards, but be obliged to report to the EPER, when its total emissions surpass the EPER levels. For Cd and compounds, the threshold value is 10 kg year^{-1}. Cd in stack emissions is bound on volatile particles. So, by reducing the emissions of volatile particles, emission of Cd can be reduced.

points of the thermal Zn refining. In 1999, remediation of dust emissions was finalized. In 2000, the emission of dust and metals in the pastilles department was handled but this department was closed in 2003, due to economic concerns. Umicore Balen was urged to design a remediation plan in 1999. Despite the introduction of filters and gas cleaning, the emissions of Cd remained problematic. After 2 more reminders from the environmental inspection, the Cd department was closed in 2002. From then, Cd measurements lie below detection limit. Filters have been installed and dust measurement has started. Dust from filters is captured in big bags and transported to other Umicore departments for further conversion (Peeters, 2006). Concerning PM, in 1996 and 1999, PM emissions in Balen lie between 10 and 20 ng m^{-3}. In 2004, this is 1 and 4 ng m^{-3} respectively in Overpelt and Balen, while in 2007 this is 0.8 and 1.8 ng m^{-3} respectively. This already comprises with the goal of 5 ng m^{-3} for 2012 (www.vmm.be).

Notwithstanding the dismantlement of the smelter in Lommel in 1974, the conversion from pyrolytic to electrolytic Zn refining in Overpelt in the 1970s, and a complete termination of the Cd production in Overpelt in 1992 and in Balen in 2002 the area remains polluted by Cd (Peeters, 2006; Nawrot *et al*., 2008).

Chapter 1.2 *Phytoremediation*

1.2.1 Theory

The residence time of metals in soils is thousands of years and they therefore create a permanent risk for human and environmental health (McGrath *et al*., 2001). There is an obvious need for remediation and risk reduction alternatives which are environmentally sound and protective of human health and the environment. Because of the vastness of the contaminated area it is impossible to apply conventional remediation techniques such as excavation and land filling, biological treatment, physic-chemical treatment (washing) and thermal desorption. Traditional techniques tend to degrade every biological activity in the soil and are expensive, according to literature between € 50 and € 560 ton^{-1} soil (McGrath *et al.*, 2001; Mulligan *et al.*, 2001). The total cost of remediating a Pb polluted soil lies around \$130-350 ton^{-1} (including research, monitoring, and excavation). The cost of phytoremediation on that same site should be about[2] \$20-80 ton^{-1} (Raskin and Ensley, 2000). Prices for conventional remediation in Belgium for 2010 were obtained from the Association for entrepreneurs in soil remediation (OVB) (Table 1-2).

Table 1-2 Minimum and maximum market prices for conventional soil remediation techniques in Belgium (January 2010)

Technique	*Min (€ ton^{-1})*	*Max (€ ton^{-1})*
Biological	20	25
Physico-chemical (washing)	25	55
Thermal desorption	40	60

Source: OVB (Association for entrepreneurs in soil remediation, Belgium), personal communication (February 2010)

Ex-situ remediation techniques include the excavation of contaminated soil, followed by a chemical or physical cleaning and return of the treated soil. *In-situ* methods are applied to stabilize the metals or to actually remove the metals. Phytoremediation is defined as “bioremediation by using plants and their associated micro-organisms, applicable for the removal or degradation of organic and inorganic pollution in soil, water and air” (EPA, 2000a; Vangronsveld

[2]Average euro dollar exchange rate in 2000 of € 1.08/\$

et al., 2009; Meers *et al.*, 2010). It uses plants to remove pollutants from the environment or to render the pollutants harmless. Within this domain, phytoextraction is defined as the use of plants for the effective removal of pollutants (metals and organics) from soil. The phytoextraction technology is not capable of full extraction of all contaminants, as only bio-available elements will be taken up by plants (Vassilev *et al.*, 2004). Subsequently, metals like Cd may be translocated and concentrated in the aerial, harvestable plant parts which are then removed by harvesting the plant (Chaney, 1983; EPA, 2000a; Vassilev *et al*., 2004; ITRC, 2009). The timetable within which the level of Cd in soil is taken to an acceptable level depends on the level at which plants take up the metals and their biomass production potential (Alkorta *et al.*, 2004; Ghosh and Singh, 2005). There is an absolute requirement for the chosen plants to accumulate elevated elemental content in harvested biomass with no loss of productivity (Maxted *et al.*, 2007b).

Phytoextraction of (heavy) metals can only be used for low to moderately contaminated soils that do not cause phytotoxicity in plants. It is also only a suitable technique when contamination is located in the upper layer of the soil, within reach of the roots, which on average is less than 50 cm (EPA, 2000a; Raskin and Ensley, 2000; Vassilev *et al*., 2004). An exception is the case for some trees, where the target zone reaches from one to several meters (EPA, 2000a; Vassilev *et al.*, 2004). The metal contamination in the Campine meets these conditions, covering a large area with moderately and superficially metal contaminated sandy soil (MIRA, 2006). Phytoremediation does not interrupt land use during and after remediation and does not affect soil fertility like conventional remediation techniques (Raskin and Ensley, 2000; Robinson *et al*., 2003). The cost of phytoremediation can be compared with regular farming activities (fertilizing, irrigation, …), except for the amendments that might be needed to increase the plant availability of the metals (Kumar *et al*., 1995; Rulkens *et al*., 1998; Suthersan, 1999; EPA, 2000a; Vassilev *et al*., 2004; ITRC, 2009).
Figure 1-1 below summarizes advantages and disadvantages of conventional and phytoremediation techniques.

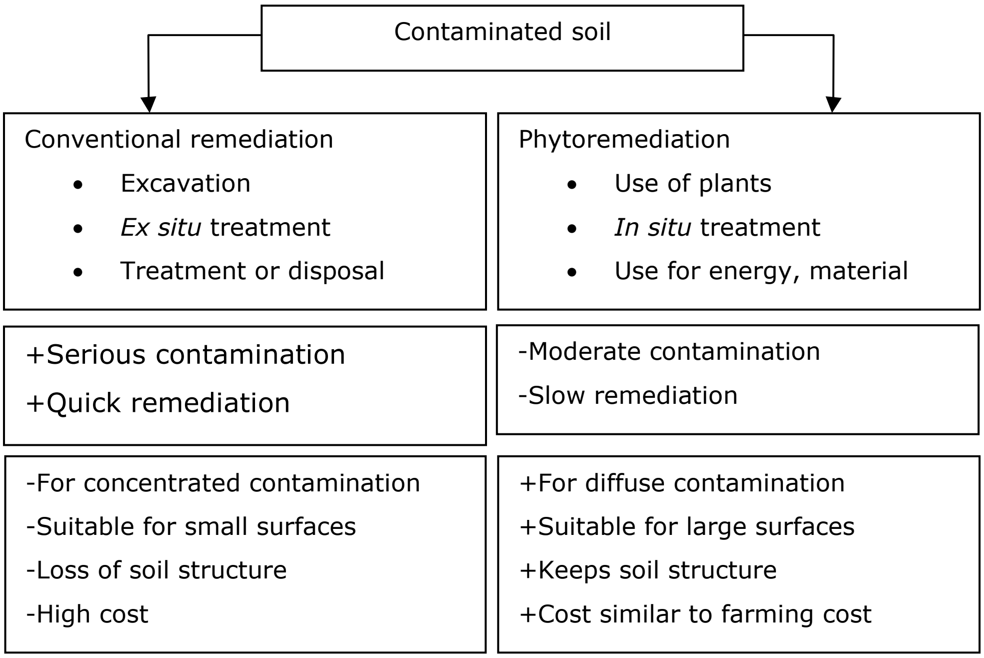


Figure 1-1 Comparison of conventional and phytoremediation technologies for metal soil remediation
Source: Witters *et al.* (2009)

1.2.2 Practice

Phytoremediation for cleanup of contaminated soil and water is often mentioned as an economically viable, effective and environmentally sustainable remediation strategy, delivering renewable energy sources, potential carbon sequestration, and improved food safety (Kumar *et al.,* 1995; Salt *et al*., 1995; Rulkens *et al.*, 1998; Susarla *et al*., 2002; Wang *et al*., 2007; Liang *et al.,* 2009; Mench *et al.*, 2010). It has already been applied on a commercial scale in the United States (Licht and Isebrands, 2005) and Sweden (Mirck *et al*., 2005)[3]. In Denmark, the phytoextraction technology with willow is amongst others tested on 2 metal contaminated sites (Jensen *et al*., 2009). To our knowledge, in Belgium, and more specifically Flanders, phytoextraction has not yet been applied on a commercial scale (Luyten, E., personal communication, July 2009).

[3]Clarinet (2002b) also mentions commercial application of phytoremediation in Ireland and France.

Research on the phytoextraction potential of plants started with hyperaccumulating plants (Salt *et al*., 1995; Chaney *et al.,* 1997). Species such as alpine penny cress (*Thlaspi caerulescens*) with natural metal accumulating characteristics were used for phytoremediation (Garbisu and Alkorta, 2001; Vassilev *et al*., 2002).

The first field experiment using natural hyperaccumulator plants was performed in 1991-1992 in sewage sludge treated plots at Woburn, UK (McGrath *et al.*, 1993). The highest Zn uptake was observed in *T. caerulescens* accumulating 2,000 to 8,000 mg Zn kg^{-1} dm in shoots when growing on soil containing total Zn concentrations of 150-450 mg kg^{-1}. The total Zn uptake was calculated to be 40 kg ha^{-1} in a single growing season. With this extraction rate it was concluded that it would take nine crops of *T. caerulescens* to reduce total Zn from 440 to 300 mg kg^{-1} – the threshold value established by the Commission of the European Community at that time. In a field trial supervised by Chaney *et al.* (1995, 1997, 1999) at Pig's Eye landfill site in St-Paul (Minnesota, USA) it was found that under optimum growth conditions *T. caerulescens* could take in Zn at a rate of 125 kg ha^{-1} $year^{-1}$ and Cd at 2 kg ha^{-1} $year^{-1}$ (Saxena *et al.*, 1999). Robinson *et al.* (1998), on the basis of both field observations and pot-soil experiments, concluded that the potential of *T. caerulescens* for Zn and Cd extraction is rather different. They reported Zn removal values very close to that observed by McGrath *et al.* (1993) and suggested that it will be not feasible to remediate the Zn contaminated mine wastes, because of both their high Zn content and low Zn bioaccumulation factor. They considered the case of Cd as different due to very high Cd accumulation in leaves of *T. caerulescens* (0.16%) originating from Ganges (France) and comparatively lower Cd contamination, especially in some agricultural soils, where phosphate fertilizers have been applied for long period.

If the high trace element concentration (Zn, Cd) of *T. caerulescens* is an advantage, its slow growth rate, low dry matter yield and rosette characteristics are main limitations (Ernst, 1998; Assunçao *et al.*, 2003). Field observations and measurements on natural populations of *T. caerulescens* have shown that these plants have an annual biomass production of 2.6 ton ha^{-1} (Robinson *et al.*, 1998). Kayser *et al.* (2000) reported a maximum yield from *T. caerulescens* of about 1 ton ha^{-1} under field trails due to poor growth and weak resistance to hot environment. On the other hand, Bennett *et al.* (1998) showed that the yield of fertilized crop of *T. caerulescens* could be easily increased by a factor of 2 - 3 without significant reduction in Zn and Cd tissue concentrations. More recently, Schwartz et al. (2003) showed evidence for this statement observing that Zn and Cd extraction by *T. caerulescens* has been improved significantly by nitrogen fertilization (80 - 200 mg N kg $soil^{-1}$). Zhao *et al.* (2003) suggested that an average *T. caerulescens* biomass of 5 ton ha^{-1} should be achieved with optimized agronomic inputs.

It is clear that the biomass potential of most hyperaccumulating species is limited, and moreover, metal accumulation is species- and even ecotype specific, and there still is a lack of knowledge concerning agricultural practices and management (Salt *et al*., 1995; Garbisu *et al*., 2001; Lombi *et al*., 2001; Vassilev *et al*., 2002, 2004; Van Nevel *et al*., 2007; Vangronsveld *et al.,* 2009; Zhang *et al*., 2009).

Therefore it was suggested that, besides accumulating reasonable amounts of metals into their above ground biomass, plants used for phytoextraction should tolerate relatively high levels of metals in the soil, while maintaining rapid growth rates and reaching a reasonably high biomass in the field (Alkorta *et al.*, 2004; Hernández-Allica *et al.*, 2008). More recently, fast-growing crops with greater biomass potential such as willow (*Salix* spp.), poplar (*Populus* spp.), maize (*Zea mays*), and rapeseed (*Brassica napus*) have been tested for phytoremediation, resulting in a final metal extraction that can be equal to hyper-accumulating plants, despite the lower metal concentrations in their tissues (Vassilev *et al*., 2002; Chaney *et al*., 2004; Van Ginneken *et al.*, 2007; Hernandez-Allica *et al*., 2008; Meers *et al*., 2005a, 2006, 2007a, 2010; Ruttens *et al.,* 2011).

If produced biomass can be valorised into an alternative income for the farmer, then the main drawback of phytoextraction, namely the extended remediation period required, may become invalid and slower working phytoremediation schemes based on gradual attenuation of the contaminants rather than short-term forced extraction may be envisaged (Robinson *et al.*, 2003). Conventional remediation techniques will take less time, but during remediation it will not be possible to validate the soil. When plant-based technologies are implemented, the repeated cropping of plants produces high amounts of biomass which need to be disposed off or better, treated appropriately to prevent any risks to the environment (Sas-Nowosielska *et al.*, 2004; Ghosh and Singh, 2005). The utilization of the obtained biomass of such a cycle as an energy resource is therefore attractive (Chaney *et al.*, 1997) and can even turn plant-based technologies into a profit making operation (Meers *et al*., 2005a).

In Europe, the production of energy maize is increasing rapidly. The biomass resulting from this crop can be applied for conversion into biogas through anaerobic digestion. As such, energy maize and biogas production represent a new branch of agriculture, which has been emerging at large scale over the past five to ten years (Meers *et al.*, 2010). Although some authors report that in comparison to other plant species, maize is a rather good accumulator of Pb (Garbisu and Alkorta, 2001; Chrysafopoulou *et al.*, 2005), other authors find that it does not actively take up trace metals (Zhang and Banks, 2006). Meers *et al.* (2005a) found that, out of four high biomass producing crops, *i.e. Brassica rapa* (rapeseed), *Cannabis sativa* (hemp), *Helianthus annuus* (sunflower) and maize, the latter exhibited the highest biomass potential on moderately metal contaminated land, with the lowest metal accumulation (Cd, Pb, Zn) in the harvestable plant parts. Although metal accumulation by maize can be chemically enhanced (McGrath *et al.*, 2001; Do Nascimento and Xing, 2006; Meers *et al.*, 2008a), remediation will still take an extensive period of time (years to decades, depending on the amount of metals that need to be extracted). Therefore, a new term for phytoremediation was proposed: "phytoattenuation" (Meers *et al.*, 2010). The main focus lies on risk reduction in utilization of metal-enriched land. This can be achieved by using crops with high biomass production potential, low metal concentration in plant parts (by using an excluder species) while allowing a maximum economic valorization of land. This is why alternative use and valorisation of the produced biomass may become a prerequisite for field-scale application of plant-based technologies as a remediation technique (Meers *et al.*, 2006).

Previous research indicated that willow can survive on metal-enriched soils (Pulford and Watson, 2003; Meers *et al*., 2007b). Its Cd accumulation potential has already been investigated widely (Landberg and Greger, 1994; Berndes *et al*., 2004; Mirck *et al*., 2005; Lewandowski *et al*., 2006) as well as its contribution to integrated brownfield redevelopment (French *et al.*, 2006). Moreover, the use of willow for energy and material purposes may contribute to the reduction of global CO_2 emissions. It can replace fossil fuels for the supply of heat, electricity and transport fuel, and can also serve as a feedstock for material production (Dornburg and Faaij, 2005). Because of the multiple environmental benefits (Borjesson, 1999a, 1999b) associated with woody crops grown on short rotations, increasing interest over the world can be noticed. Willow has several characteristics that make it ideal for short rotation cycles, including high yields obtained in a few years, ease of vegetative propagation, a broad genetic base, a short breeding cycle, and the ability to resprout after multiple harvests (Kopp *et al*., 2001; Volk *et al*., 2004). Its full environmental potential (Kuzovkina and Quigley, 2005) and economics have already been studied in Sweden (Rosenqvist *et al*., 2000), Ireland (Rosenqvist and Dawson, 2005), Finland (Tahvanainen and Rytkonen, 1999), Poland (Ericsson *et al.*, 2006), Germany (Scholz and Ellerbrock, 2002), United States (Adegbidi *et al*., 2001; Keoleian and Volk, 2005) and the UK (Mitchell *et al*., 1999). The use of willow as an income generating crop by agriculture is not mentioned in agricultural statistics in Belgium up to 2005 (FOD Economy: statistics, 2006). However, experimental plantings have occurred recently (Van Ginneken *et al*., 2007; Ruttens *et al*., 2008; Meiresonne *et al*., 2009). SRC of poplar has been indicated by the Institute for Nature and Forest as having the right characteristics to serve as a remediating crop (Meiresonne, 2006). Moreover, calculations on the energy potential of willow in Belgium seem promising (Cidad *et al*., 2003).

Recently, fast-growing high biomass crop plants that accumulate moderate levels of metals in their shoots are being tested for their metal phytoextraction potential. Rapeseed (*Brassica napus*) has been type casted as an above average accumulator of metals (Bernhard *et al*., 2005; Selvam and Wong, 2009; Shi and Cai, 2009). Therefore, crops from the *Brassica* family have already been suggested for phytoremediation (Marchiol *et al*., 2004). Field experiments confirm that certain rapeseed crops are suitable for phytoextraction of moderately metal contaminated soils. Moreover, crops from the *Brassica* family display a significant metal tolerance (Bernhard *et al*., 2005; Hernandez-Allica, 2008; Shi and Cai, 2009). Grispen *et al*. (2005) conducted a screening for natural variation in Cd accumulated by 77 *Brassica napus*. This yielded potential candidates for phytoextraction in agricultural practice. After harvest, rapeseed results in rich oil containing seeds, and straw. The seeds could be sold for biodiesel production. The use of rapeseed as an income generating crop in Belgium occupies almost 11,000 ha in Belgium in 2007, most of it being grown in the Southern part of Belgium (FOD Economy: SMEs, independent Professions and Energy, personal communication, March 2009).

Chapter 1.3 *Economic decision theory*

In general, the essence of a decision is considered as a choice between alternatives. A distinction can be made between prescriptive or normative decision theory (*i.e.* prescribing what a decision maker should do) and descriptive decision theory (*i.e.* describing what a decision maker actually does). In descriptive decision theory, which is concerned with understanding and predicting how people actually reach decisions, the rational behavior of a human decision maker is considered as subjective and bounded[4]. Prescriptive or normative decision theory optimizes the choice between the different alternatives using mathematical functions (Merkhofer, 1987). Decision science has not yet developed a universally accepted methodology free from criticism for analyzing social decisions involving risk. Cost-benefit analysis (CBA) is a useful and popular tool, aside from cost-effectiveness analysis, multi-criteria (decision) analysis, risk assessment and environmental impact assessment (Kotchen, 2010).

1.3.1 Cost-Benefit Theory and Cost-benefit analysis

1.3.1.1 General

Any possible decision is based on the mathematical solution of a function that integrates different criteria. The difficulty lies in determining this function so that it reflects our main intentions. Typically, (environmental) decisions are based on an analysis using a CBA framework. Commonly, the Net Present Value (NPV) of a stream of incoming and outgoing cash flows is maximized (Munda, 1996; Mirasgedis and Diakoulaki, 1997; Crookes and De Wit, 2002; Ding, 2005).

Cost-Benefit Theory (CBT) is the basic theory underlying CBA. It is based on the concept of rationality of utility maximization. Human beings can be represented as separate, autonomous individuals seeking to satisfy preferences of varying

[4]The term bounded rationality has been introduced by H.A. Simon in 1957 in his book 'Models of man'. Here, he states that the capacity of the human mind for formulating and solving complex problems is very small compared with the size of problems whose solution is required for (objectively) rational behavior in the real world -or even for a reasonable approximation to such (objective) rationality. This has lead to models of human decision making in terms of satisfying rather than optimizing.

importance to them. The means needed to satisfy these preferences are limited (*i.e.* individuals act under the condition of relative scarcity), and these scarce means are capable of alternative application. What individuals then do is maximize their different preferences given the scarcity of means, and each individual makes decisions based on a comparison of costs and benefits. CBT is then concerned with the maximization of the aggregate value of goods and services consumed by individuals.

To value individual preferences, the concepts of willingness to pay (WTP) and willingness to accept (WTA) are applied: an individual's valuation of impacts is what he or she would be willing to sacrifice to incur or avoid them[5]. The adoption of the WTP principle implies the use of market values for valuing marketable commodities. WTP can be found on the individual demand curve, while consumer surplus is found on the aggregate demand curve, defined as the collective WTP. To aggregate values to the social level, individual values, regardless of to whom they occur, are added to yield a measure of the total social utility of the alternative.

1.3.1.2 The pursuit of efficiency

The central premise of CBT is that alternatives are ranked according to a systematic comparison of advantages (benefits) and disadvantages (costs) that result from the estimated consequences of the alternative. Thus, the theory does not involve the concept of a social decision maker with special responsibility for the decision. On the contrary, individuals are assumed to be the appropriate judges for valuing consequences. A best alternative is then defined in terms of efficiency, *i.e.* it maximizes total surplus to society (Merkhofer, 1987; Hanley, 2000).

CBA approaches strive to implement the efficiency criterion of CBT by investigating whether the consequences of an alternative are likely to increase

[5]WTP and WTA are not necessarily equal, as their starting points differ. WTP uses the level of utility without improvement as a reference point, while WTA uses the level of utility with improvement as a reference point.

efficiency. NPV functions as a decision criterion for maximizing efficiency, under the assumption of a perfect market. Other decision criteria are the Internal Rate of Return, or the Benefit Cost Ratio.

NPV is found by multiplying benefits and costs at (the end of) each year t by a time dependent weight. As all effects will not occur in one year, but over several years (t: 0,…n) and as people prefer one unit today rather than tomorrow, time preferences are taken into account, and yearly benefits and costs are discounted using a discount rate (i). A capital sum in the initial year of investment will not have to be discounted, except when it is converted into an annual capital cost. To account for inflation, real or nominal values can be used, as long as they are used consistently. Option 1 is then preferred over option 2 if (Eq. 2) holds[6].

$$NPV= \sum_{t=0}^{n} (B_t - C_t)/(1 + i)^t \qquad \text{(Eq. 1)}$$

$$NPV_1>NPV_2\ (>0) \qquad \text{(Eq. 2)}$$

1.3.1.3 Underlying assumptions of CBT

CBT has some important underlying assumptions (Merkhofer, 1987; Munda, 1996; Hanley, 2000; Boardman *et al.*, 2006; Kotchen, 2010). First, the theory considers benefits and costs that occur to the total social system, it does not consider separately the direct benefits to intended beneficiaries and direct expenditures by the implementing party. Second, the theory assumes that each individual maximizes its total utility and that the invisible hand automatically implies a maximized total surplus. As a consequence, authority for value judgments is decentralized to all stakeholders in the decision. Third, it is implicitly assumed that the relationship between monetary value and welfare is linear, *i.e.* an individual with twice the amount of money will be twice as well off. Fourth, individuals can be compensated with money for any impact, at the

[6]An alternative indicator commonly used is the Internal Rate of Return (IRR). IRR can be calculated as the i for which NPV=0. An investment is then accepted when IRR≥I, I being the required return rate. Option 1 is then preferred over option 2 if $IRR_1>IRR_2$ (≥i). The benefit cost ratio is the ratio of discounted benefits to discounted costs and the decision rule is to proceed when this ratio ≥ 1 (Hanley, 2000).

private and societal level. Thus resources do not have an intrinsic value; their value is derived from their consumption. A fifth assumption (also the most common source of critique towards the theory) is the fact that the theory disregards income distribution or equity[7]. This is implied because market prices and WTP depend on individuals' wealth as well as on their preferences. Therefore, accepting existing market prices requires accepting the existing distribution of wealth, as an individual's influence on prices is proportional to the individual's wealth. A final assumption is the potential compensation principle or potential Pareto principle. The Pareto principle states that a decision is an improvement if it makes one or more persons better off without simultaneously making at least one person worse off. CBA is based on the potential Pareto principle or Kaldor-Hicks principle. A potential policy will pass the CBA test when the winners from the policy are able to compensate the losers. It does however not say, that they should compensate.

Due to the construction of the single decision criterion (NPV), it is mathematically possible to find an optimal solution. As mentioned, the criterion of social desirability is usually expressed in terms of the potential Pareto criterion. Thus, the basic theorem of welfare economics is to legitimize rational behavior as being socially desirable and to justify government intervention to improve conditions under which individuals make choices. Government intervention is appropriate when market failures, such as described below, exist, when the invisible hand is not maximizing social welfare (Pearce and Turner, 1990).

1.3.2 Market failures/Missing markets with environmental consequences

In a perfectly competitive market, prices have three functions that guide consumers and producers up to the point that maximizes social welfare. First,

[7]Fullerton and Stavins (1998) find in the considerable dispute surrounding distributional equity a reason for the fewer studies on this subject and agree that the combination of efficiency and distributional equity in one analysis has not been studied yet in a satisfactory manner.

they allocate goods to agents that value them most highly. Second, they inform on the relative scarcity of goods. Third, they provide incentives to use more or less of a good and to move towards the point of maximum efficiency (Graves, 2007). In a perfect market, a producer will supply goods as long as marginal production costs for an extra good lie below the price paid for the good. A consumer will continue to buy a good as long as its marginal benefit exceeds the price. The self-interest of both consumer and producer leads, guided by the invisible hand, to an equilibrium market price and an equilibrium quantity of goods exchanged. In a system with well defined property rights and competitive markets, producers and consumers maximize their private surpluses. The price system then induces the self-interested parties to make choices that are socially efficient. Therefore, in a perfect market[8], government intervention would not improve social welfare. The decision analysis as described above will only lead to a completely efficient allocation of resources within a perfect market. However, if the market for a commodity is distorted in any way, prices might not reflect an individual's WTP. The consequences of these circumstances will be the focus of the next paragraphs. In our case, under what conditions does economic analysis not straightforwardly lead to the most socially efficient outcome? Or, under what conditions can government intervention lead to an increased social welfare? To answer this question, we need to define the concept of property rights (Tietenberg, 2003).

A property right is a bundle of entitlements defining the owner's rights, privileges and limitations for use of the resource. An efficient structure of property rights has three characteristics. Exclusivity points to the fact that all benefits and costs that result from using or owning a resource should accrue to the owners alone. Transferability of the rights should exist between owners on a voluntary basis. Enforceability means that there is no intrusion by others. When these conditions are fulfilled, an agent that holds a property right on a resource

[8]In a perfect market there are no public goods, no externalities, no monopoly buyers and sellers, no increasing returns to scale, no information problems, no transaction costs, no taxes, no common property and no other distortions between costs paid by buyers and benefits received by sellers (Fullerton and Stavins, 1998).

will use this resource the most efficient as possible because a loss in value of the resource is a personal loss (Tietenberg, 2003).

Inefficient outcomes occur when the market structure is imperfect and when property rights are not properly defined, resulting in externalities and public goods. By some, these are called market failures, others prefer the term missing markets, and missing markets always lead to resource misallocation. Externalities occur when the condition of exclusivity is violated, leading to an output of a given commodity that is too large. Public goods occur when in the absence of exclusivity each person is able to become a free rider on another person's contributions (Tietenberg, 2003; Graves, 2007). We shall discuss these missing markets/market failures in detail in the next paragraph.

1.3.2.1 Public goods

In a perfectly competitive market, rational producers and consumers will produce private goods up to the level where marginal benefits equal marginal costs, maximizing producer and consumer surplus respectively. If there would exist such a perfectly functioning market for public goods, the same conclusions could be made for public goods. However, a public good is a good from which the consumption collectively affects multiple individuals. Two characteristics distinguish a public good from a private good. First, a unit of a public good can be consumed by many (non-rivalry). Unlike an ordinary good, one person's satisfaction from a public good is not diminished by the satisfaction gained by others. Second, a public good is available to everyone (non-exclusivity), it is almost impossible to exclude someone from using the good (Johansson, 1987; Merkhofer, 1987). In reality there is a whole range of goods, ranging from pure private ones to pure public ones. So, a public good is a special case of a missing market. Unlike the case of externalities of private goods where the market does not reflect environmental costs or benefits, for public goods there is no market at all. What economists then do is try to ascertain where the marginal benefit curve lies. The marginal cost curve is considered relatively easy to identify (Graves, 2007).

1.3.2.2 Externalities

Vatn and Bromley (1997) state that the term market failure is an unfortunate choice as externalities are a rational result of the market as such, *i.e.* the absence of properly defined property rights are common to most markets. An external effect occurs when the utility of one consumer is affected by the consumption or production of another consumer or producer (Johansson, 1987). Or, an externality exists if the production or consumption of a product generates costs or benefits to others which are not reflected in prices. The prices that consumers face do not reflect true social values.

According to economic theory, an individual whose initial desire for a commodity exceeds its price will continue to purchase the commodity until the benefit derived from the last amount purchased equals the price paid for that amount. When externalities are present this will lead to an over- or underconsumption and over– or underproduction of the good, leading to a total social surplus that is diminished with a dead weight loss. As a result, the free market does not produce an efficient level of welfare (Merkhofer, 1987; Graves, 2007). An example of an externality can be found in the potential CO_2 abatement of biomass based energy. The external effect might not be correctly reflected in the price of biomass if there is no (appropriate) policy. So, when externalities exist, government may be justified in intervening to force a level of welfare that is more socially desirable than the inappropriate one reached through the market. One way for government to change human behavior is through the implementation of mandatory standards and regulations or market oriented incentives designated to force individuals to take actions that lessen risk (Merkhofer, 1987).

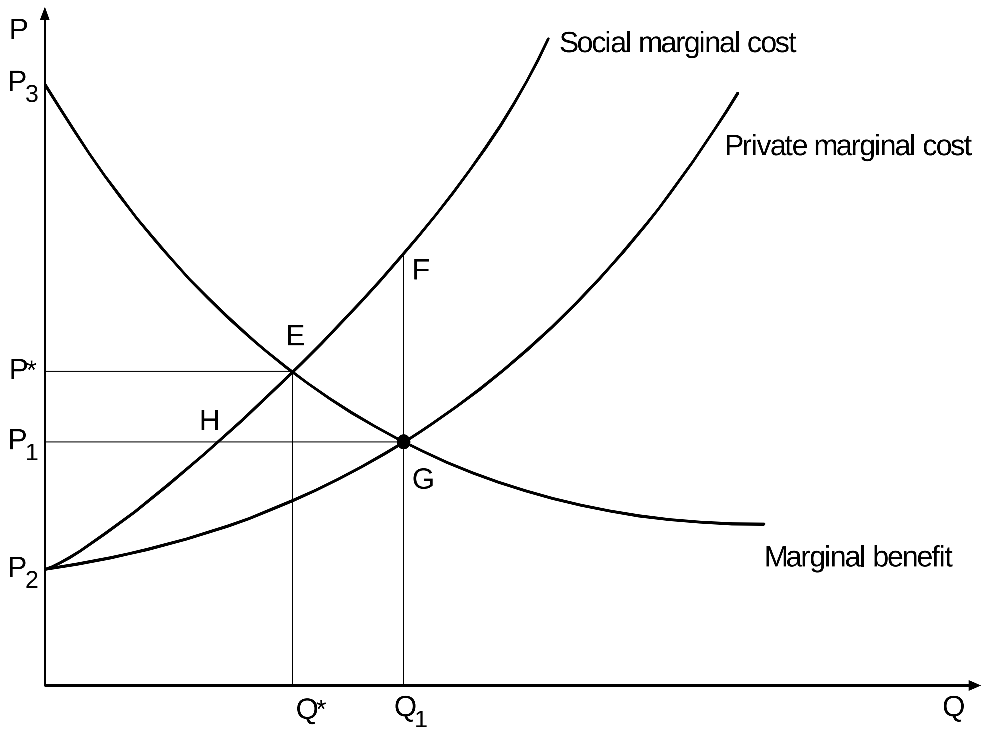


Figure 1-2 Illustration of an externality, with Q_1 the quantity of the good or service produced and consumed at price P_1 before internalization of the externality, and Q* and P* respectively the quantity and price after the externality has been internalized
Source: based on EPA (2000b)

The situation before policy implementation results in marginal revenue and private marginal costs intersecting at price P_1, leading to consumption of private good Q_1 (Figure 1-2). The producer surplus at this point is P_1GP_2, consumer surplus is P_1GP_3. Because of the (negative) externality, there is a social damage of P_2FG (including a dead weight loss of EFG). The net social welfare is then P_2EP_3 – EFG. Through introduction of the correct policy, the optimal Q* is now determined by the intersection of the marginal revenue curve with the marginal social cost curve. The producer surplus is now P_2EP^*, whereas consumer surplus is P^*EP_3. The net social welfare is now P_2EP_3. Although there is a decrease in producer- and consumer surplus, the overall social welfare has increased because of the reduction in external costs. There is no longer a dead weight loss due to inefficient consumption and production of a good.

Another way to demonstrate this is through the interdependence between producer functions, between consumer functions, and between consumer and producer functions which cause externalities (Castle, 1970). When we use production relationships to illustrate externalities in a formula: $Y_A = F(X_1, X_2, \ldots, X_n)$ and $X_n = F(Y_B)$, with Y_A the output of producer A, Y_B the output of producer B, and $X_1, X_2, \ldots, X_n$ inputs, and suppose X_n is a negative input such as CO_2 emission.

$\partial Y_A / \partial X_n < 0$ and $\partial X_n / \partial Y_B < 0$ or $\partial Y_A / \partial X_n > 0$ and $\partial X_n / \partial Y_B > 0$ (Eq. 3)

In (Eq. 3) external economies (positive spill-over or external benefit) exist. The production of Y_B reduces the production of X_n and this reduction in X_n will lead to a positive effect on the production of Y_A or the production of Y_B augments the production of X_n and this augmentation in X_n will lead to a positive effect on the production of Y_A.

$\partial Y_A / \partial X_n < 0$ and $\partial X_n / \partial Y_B > 0$ or $\partial Y_A / \partial X_n > 0$ and $\partial X_n / \partial Y_B < 0$ (Eq. 4)

In (Eq. 4) external diseconomies (negative spill-over or external cost) exist. The production of Y_B augments the production of X_n and this augmentation in X_n will have a negative effect on the production of Y_A or the production of Y_B reduces the production of X_n and this reduction in X_n will have a negative effect on the production of Y_A.

Lofgren (2000) distinguishes between private and public externalities. Many externalities have the character of a public good, they have an effect on all of us and the consumption of the externality by one of us does not lead to less consumption by others. Again, we can take the example of CO_2 emission by burning fossil fuel. The emission affects all of us and by burning biomass based fuel, we avoid these emissions. Therefore we can say that the CO_2 abatement by producing energy from biomass is an externality with a public character. Metals in rest products from biomass based energy production could be defined as examples of private externalities[9]. If the rest products after energy production are not managed properly, metals might *e.g.* end up on the land again and be recycled in the food chain, affecting local people. This distinction between private and public externalities is important up to the point that it might give an indication of the extent to which policies are easier to implement. Intuitively, it seems easier to develop and integrate a policy that handles local, private externalities. However, the general underlying assumption of social welfare maximization is indifferent between private and public external effects and we will not distinguish between them.

Policy instruments to internalize externalities require measuring the damage done or benefits caused by the production and consumption of environmental goods and services. Since environmental problems are very often caused by the fact that they are not valued appropriately, the valuation of environmental goods should take a prominent place in the discussion of these policies.

1.3.3 Valuation techniques

In internalizing externalities, policies should be developed such that the externalities are internalized in a way that the benefits of internalizing (*e.g.* surface under marginal damage costs, and between the suboptimal level of emission and optimal level of emission) exceed the costs of abatement (*e.g.* surface under marginal abatement cost curve and between the suboptimal level of emission and optimal level of emission). This will be explained in detail in 1.3.6. The point that we are making here is that we should put an economic value on costs and benefits. How this could be done, is explained below.

The NPV decision rule can only be used under the assumption of strong commensurability. A set of conversion factors should therefore exist to transform all outcome variables underlying a given action into a single

[9]We are assuming here that the (elevated) metal concentration in biomass is not an objective property of biomass, and is thus a potential externality. Objective properties of biomass are dry matter content, water content, energy level, … Arguments for the contrary approach could be understood (Mendelsohn, R., personal communication, March 2011).

composite measure (Munda, 1996). This can be achieved by calculating the Total Economic Value (TEV) as the sum of Use Value and Non-use Value (Figure 1-3). The use value is the utility of using a good or service and can be direct (sum of consumptive and non-consumptive value), indirect (through using another good or service), or optional (possibility of using it in the future). The non-use value is the raise in utility of a good for an individual, without that individual actually using the good. It results from the fact that people wish to maintain or improve environmental assets out of sympathy for *e.g.* animals and nature, it is an intrinsic value. It is the sum of bequest value (concern for future and generations) and existence value (there is no past, present or future use, the value is simply based on knowing that the good exists). An example is the fact whether a forest will still be available in the future (Krutilla, 1967; Markandya *et al.*, 2002; Randall, 2002; Ruijgrok *et al*., 2004; Kotchen, M., personal communication, September 2009).

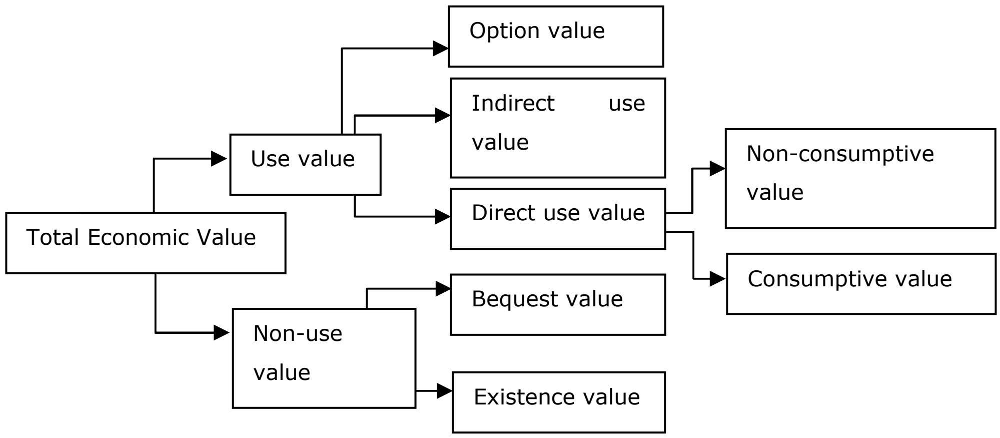


Figure 1-3 Total economic value
Source: based on Ruijgrok *et al.* (2004); LNE (2007); Kotchen, M., personal communication, (September 2009)

In CBA, weighting different decision criteria is based on the monetary value placed on the different criteria. Like equilibrium prices for goods traded on the market are assumed to balance aggregate demand with aggregate supply, to maximize social efficiency, likewise values for non-market goods, established through the techniques described below, will reflect aggregate equilibrium values, maximizing efficiency.

On the one hand[10], market methods can be used when valuing environmental services and goods which are directly traded as a market commodity. The input-output method gives an overview of the amount of money that flows between different groups and relates this to the existence of an environmental good. The market price method (also referred to as production factor/production function method) is a direct method that determines the WTP by the market price for the good. It determines the economic value of an environmental good based on the physical impact of this good on the economic production process of a given private market good. Thus in our case, biomass yield reduction due to metal pollution can be valued at the price of the crop, assuming that the price is the same as in the reference situation. The great value of this method is that it relies directly on existing market prices. However, if the decision action produces substantial changes in the market, then this must be taken into account when using market prices.

On the other hand, environmental goods and services have public good characteristics that make it difficult for markets to function well. Non-market valuation is a technical term used to refer to a set of valuation techniques of goods and services which are not represented on the market and do not have a market price. Generally, non-market valuation methods are classified in expressed and revealed categories (Bickel and Friedrich, 2005). Revealed preference methods use data on actual choices made by individuals in related markets, while expressed methods base their analysis on presenting consumers with hypothetical choices. The contingent valuation method is a stated/expressed preference method that creates a hypothetical market for an environmental good. By questioning the target group, one can derive its WTP for a change in the given service/good. The intuition behind this method is that if the questionnaire is designed with care, people can provide reliable evidence of

[10]Other authors distinguish between demand curve and non-demand curve approaches (Longo *et al.*, 2008). The classification used here is offered to facilitate the discussion and is similar in many aspects to Bickel and Friedrich (2005), but alternatives are offered by other researchers and have been included as much as possible. Our classification is not intended to be comprehensive or extensive.

values of goods and services. Literature on the advantages and disadvantages of the technique are large. Choice experiments introduce a set of alternatives regarding the situation concerning an environmental good/service to the respondents, who are then asked to rank the alternatives. The ranking of choices leads to the determination of the marginal rate of substitution between any characteristic and the environmental good/service. The averting behavior method is a revealed preference method that studies the change in behavior/expenditures caused by an environmental problem. In the case of remediation this might *e.g.* be the cost for reducing pollutants causing the damage, the actions people take so as not to enter into contact with the pollutants. The travel cost method relates differences in travel expenditure to differences in visits to an area. This information can then be used to infer a demand function for the non-market good/service. From this the consumer surplus and thus the economic value of *e.g.* that area can be determined. The hedonic pricing method is a revealed preference method that estimates the value based on behavior on the market of a related good or service. It estimates implicit prices for a market commodity when an environmental good/service is viewed as an attribute of this commodity. The method measures the marginal value created by the activity/action. It is mainly used for properties and wages. Clauw (2007) applied the hedonic price method to valorize the impact of contamination on private house prices. The substitution cost method estimates the costs made by a government or a company to restore or replace an environmental good. It assumes that the economic value of a non-market good can be estimated by the market price of a substitute market good (*i.e.* one that can replace the non-market good). The advantage lies in the direct use of market prices. However, arguments against arise because the method assumes that the cost of replacement equals true social values and this doesn't have to be the case (Merkhofer, 1987; Markandya *et al.*, 2002; EPA, 2000b; Ruijgrok *et al.*, 2004; Bickel and Friedrich, 2005; Lewandowski *et al*., 2006; LNE, 2007).

1.3.4 Alternatives to CBA

Most economists would argue that economic efficiency ought to be one of the fundamental criteria for evaluating regulations on environment. Society has a limited amount of resources, and CBA is able to explicitly define the trade-offs to make the decision or regulation that will lead to the highest social surplus. Arrow *et al.* (1996) suggest that although CBA should play an important role in environmental decision making, it should not be the sole base.

In literature on CBA and environmental protection we find objections[11] on the use of a condensed single objective in the evaluation process since it is considered insufficient when taking environmental values into account (Hanley, 2000; Crookes and de Wit, 2002; Ding, 2005; Madlener *et al.*, 2009). The use of CBA valuation techniques has its drawbacks. There are growing concerns that values of environmental goods and services are ignored and underestimated (Ding, 2005). According to Samuelson (1954), *"it is in the selfish interest of each person to give false signals, to pretend to have less interest in a given collective consumption activity than he really has".* Moreover, NPV is an additive value function, where different dimensions are condensed by using a simple linear weighting rule, yielding a total value. In the case in which environmental dimensions are involved, the use of such a linear aggregation procedure then implies that among the different ecologic, social and economic aspects there is no interdependence. Munda (1996) talks about monetary reductionism. Fullerton and Stavins (1998) refute this by saying that economists (i) are not only considered with the financial value[12] of goods and services, and (ii) rather see monetary values as a functional method to add disparate values. Finally, CBA assumes full compensability. It is thus always possible to find an amount of money for environmental quality improvements or for environmental quality deterioration that keeps utility high. As a result, the use of CBA offers no guarantee for sustainability, according to Munda (1996). We will build further on this when we define sustainability (see 1.3.5). Suggestions are made that CBA could and should be supplemented with a technique to measure environmental

[11] Kelman (1981) gives an extended critique of CBA based on ethical principles and explains for example why in the area of environmental regulation a certain decision might be right even though benefits do not exceed costs. Crookes and de Wit (2002) point to ethical questions that are raised to certain valuations and the low public and political acceptability of economic models for valuing the environment.

[12] There is a difference between financial profitability and economic efficiency, where financial value is foremost interested in the private costs and benefits, economic analysis optimizes social value (Turner, 2000).

costs in other than monetary terms (Joubert *et al*., 1997; Mirasgedis and Diakoulaki, 1997).

Cost-effectiveness analysis (CEA) circumvents three common constraints of CBA (Markandya *et al*., 2002; Brent, 2003). It is an appropriate technique when it is not possible to value an impact, when analysts recognize that not all impacts can be monetized and/or integrated, and when the linkage between a good and the final objective is not clear. It compares alternatives based on the ratio of their cost (C) and a quantified but not monetized effectiveness measure (E). CEA bases its conclusion on the fact that one project can achieve an effect with fewer resources than another project. Therefore, it cannot ascertain whether choosing even the best project is in itself socially worthwhile, *i.e.* B≥C. Moreover, it cannot compare projects that compare different kinds of effects. Whether an effect is regarded as a positive consequence on the denominator of the C/E ratio or is treated as a cost saving and hence deducted from the nominator, makes a difference. Cost utility analysis, cost minimization, risk assessment, multi criteria decision analysis, and environmental impact assessment are other alternatives.

Multi Criteria Decision Analysis (MCDA) is the term used to include all methods that incorporate multiple criteria when helping decision makers in their problem solving. Criteria are defined as measures of performance by which alternatives are judged. A final score for each option is calculated based on aggregation of performances on all criteria and weights for each of the criteria. This phase depends heavily on the technique by which the weights are derived, and the aggregation method (Lahdelma *et al.*, 2000; Diakoulaki and Grafakos, 2004; Løken, 2007). MCDA has already widely been used to assess forest management (Ananda and Herath, 2008), sustainability of agricultural crops (Zander and Kächele, 1999; Dogliotti *et al.*, 2004; Sadok *et al*., 2008), contaminated ground water management (Khadam and Kaluarachchi, 2003) and contaminated land management methods (Janikowski *et al.*, 2000). However, to the best of our knowledge, it has not yet been used to assess crop choice for contaminated land management.

Why would one prefer to apply a MCDA method over a CBA analysis? The first concept of relevance in the CBA versus MCDA discussion is commensurability. Strong commensurability indicates that there exists a common measure of the different consequences of an action based on a cardinal scale of measurement (*e.g.* monetary terms in CBA). Weak commensurability indicates that there only exists a common measure based on an ordinal scale of measurement. Strong commensurability has to be presupposed in a CBA setting, while for MCDA this is not a necessity. A second concept is *comparability* of values, where strong comparability means that there exists a single comparative term by which all different actions can be ranked (*e.g.* NPV in CBA). Weak comparability means that one has to accept the existence of conflicting objectives and is considered the foundation of multi criteria evaluation (Munda, 1996; Diakoulaki and Grafakos, 1997). Third, criterion "a" is preferentially independent of any criterion "b" when preference scores can be assigned for the alternatives on criterion "a" without knowing what the alternatives' preference scores are on criterion "b". If this is true for all criteria regarding each one of the remaining criteria, the criteria are considered mutually preference independent (Munda, 1996; Keeney, 2006). This is of importance for linearly adding up scores on different criteria, as is the case for CBA and some MCDA methods. Fourth, when we aggregate several dimensions we need to take into consideration the concept of compensation. Compensability refers to the existence of trade-offs, *i.e.* the possibility of offsetting a disadvantage on one criterion by a sufficiently large advantage on another criterion. In CBA, losses in one impact can be fully compensated by gains in another. The theoretical model adopted in CBA is the potential compensation principle. This principle declares that a policy should be adopted when those who gain from the policy could compensate those who lose and still have some gains left over (Munda, 1996; Boardman *et al.*, 2006). A MCDA method can be compensatory (*e.g.* weighted sum), non-compensatory (no trade-off, *e.g.* lexicographic method) or partially compensatory (most MCDA methods) (Munda, 1996; Guitouni and Martel, 1998). Advantages of MCDA methods are the ability to include monetary and non-monetary measures of objectives, and to include a wide range of objectives. However, MCDA techniques are not supported by a coherent theory of social welfare, weights do not represent consumer preferences and depend on the decision maker, and often are black boxes as they generate one number and seem complicated to the outside world (Hanley, 2000).

1.3.5 Does CBA guarantee sustainability

1.3.5.1 Sustainability

The concept of sustainability can be broken down into 2 components: intergenerational equity and dynamic efficiency. Intergenerational equity on the one hand emphasizes the ability of the economy to maintain living standards, *i.e.* intertemporal[13] social welfare (V_t) should <u>not decrease</u> over time. This is the concept as embraced in the Brundtland Report. Arrow *et al.* (2003) demonstrate that this means that the accounting value of all society's capital changes, including natural capital, and manufactured, human, knowledge capital is not negative at time t. The wideness of the concept of capital implies that it is not necessary to keep a specific set of resources at time t. Resources are substitutable, suggesting that we do not owe the future any particular good or any particular natural resource. What we are obliged to leave behind is a generalized capacity to create well-being. If this principle is accepted, we end up in the world of substitution and trade-offs. However, it stays rational and logical to want to preserve a particular environmental good or service, but not under the heading of sustainability. So when we use up something that is irreplaceable, we should provide a substitute of equal value. The "something" that we should provide could *e.g.* be technology[14] (Solow, 1991; Arrow *et al.*, 2003). Current generations are thus making investments necessary to make high levels of future consumption possible.

Dynamic efficiency on the other hand focuses on <u>maximizing</u> V_t (at every time period t), *i.e.* the integral of discounted values of current and future utility from society's aggregate consumption from time t to infinity. In maximizing this V_t,

[13]Intertemporal welfare deals with the distribution of welfare between the current time and the future, whereas intratemporal welfare deals with the distribution within one time period, the equity issue (Arrow *et al.*, 2003).
[14]This is related to the concept of dynamic efficiency.

there is an optimal consumption path to be followed[15]. Current consumption is excessive when it lies above the level of current consumption prescribed by this path. Or, current consumption is excessive if lowering it and instead increasing investment would increase future utility more than the decrease in current utility.

An economy is sustainable if and only if it is dynamically efficient and if the stream of maximized welfare is not decreasing over time. This is a rather demanding definition and is hard to be achieved by public policy. Economic maximization does not necessarily lead to sustainability. In defining sustainable development, there is no presumption that the consumption path is maximizing V_t at time t. But also the other way around, a policy that passes the social CBA test could result in a decrease in V_t in time. Although dynamically efficient allocations do not automatically satisfy intergenerational criteria, *i.e.* future generations having the same welfare as current generations, they can be perfectly "consistent" with intergenerational equity, even in an economy relying on exhaustible resources. This would require a degree of investment from the first generation (Tietenberg, 2003).

Much in the same way as we apply the Kaldor-Hicks criterion instead of the Pareto criterion to judge whether a policy is Pareto improving, we can think of an economy as becoming sustainable if it fulfills the criterion of dynamic efficiency (Stavins *et al.*, 2002). It can then be made sustainable by intergenerational transfers. The Hartwick rule suggests that one way to tell whether an allocation is sustainable is to examine what is happening to the principal amount of capital in time. If this amount of capital declines, we consume too much. If it increases or remains stable, our current consumption is sustainable. Hartwick demonstrated that a constant level of consumption could be maintained in the future if the scarcity rent was invested in capital, *i.e.* the cost of increased scarcity from depletion of an exhaustible resource should be offset by this investment.

[15]This is theoretically true in a perfect market. In an imperfect market, we will have to correct prices for *e.g.* external costs to arrive at the optimal consumption path. But, as mentioned before, this is not as easy and straightforward as it seems.

Current environmental protection tries to withhold us from burdening the environment, from free-riding on the future. It is basically a problem of savings and investment (Solow, 1991). In this respect, it can be argued whether the goal of sustainability could be left entirely to the market (Solow, 1991; Arrow *et al*., 2003). There is a dual connection between environmental issues and sustainability. On the one hand, the environment needs protection by public policy because by damaging the environment we can profit and at the same time have (some of) the costs borne by others, by the future. On the other hand, sustainability is a problem precisely because its definition implies and we all know that we can make profits at the expense of the future (free-riding) (Solow, 1991).

Therefore, correct environmental policy setting is important. Standard policy remedies for improving only economic efficiency do not guarantee sustainability. At the same time, such policies do not necessarily have to conflict with the sustainability concept (Arrow *et al*., 2003). Of course, we will make mistakes designing policies. We will attribute to the future wrong tastes and excessive or undervalued technological capacities. Ecologists often argue that certain natural resources are undervalued because economists are too optimistic about substitutes for these resources. When natural resource inputs are priced below social cost, the overall level and composition of consumption can lead to excessive natural resource use (Arrow *et al.*, 2003). On the other side, it is still possible to perform the social CBA to see whether a policy at least increases intertemporal social welfare (as the over- or underestimation of benefits and costs is consistent, and the analysis is marginal). Also, the determination of the discount rate is open to debate and will have an impact on the optimal consumption path. However, all these considerations should not abstain us from making policies, the guesswork has to be done, as we will choose policies to avoid potentially catastrophic errors (Solow, 1991, 1992; Arrow *et al*., 2003).

1.3.5.2 Discounting and the choice of a social discount rate

In public policy evaluation, social discounting is perceived as a way to evaluate policy in an overall social perspective. The conceptual foundation of accounting is based on the fact that present consumption differs from future consumption. NPV is then the resulting representative of social value that can be used in assessing environmental policies.

When benefits and costs occur in the far future, the choice of a discount rate has a large impact on the final outcome: the higher the discount rate, the less weight is placed on the costs and benefits occurring in the future. This implies that society cares less about what happens in the future as a result of current action. There are two ways in which a social discount rate can be derived: intra-generational and intergenerational discounting. Intra-generational discounting does not explicitly involve extremely long time horizons and can be based on the opportunity cost of capital, *i.e.* the marginal rate of return on investment. Intergenerational accounting covers very long time horizons involving multiple generations and can be based on the social rate of time preference (SRTP). In theory, in an ideal economy (efficient markets and no taxes), these result in the same discount rate. However, in the real world a gap between the two exists (Azar and Sterner, 1996; EPA, 2000b).

$$SRTP = \xi g + t \qquad \text{(Eq. 5)}$$

SRTP (Eq. 5) consists of two components[16]: the expectation that we will be richer in the future (ξg), and pure time preference (t). The first component is based on the decreasing marginal utility of consumption. ξ is the percentage fall in the additional utility derived from each percentage increase in consumption[17], and g is the growth rate of per capita consumption. If income is expected to grow over time, the assumption of decreasing marginal utility implies that an additional unit of consumption at time t is worth less than if it had been consumed today. The second component is rationalized in terms of impatience and uncertainty about the future existence of humankind, utility today is considered better than utility tomorrow. We discount a cost or an income solely due to its position in time, no matter how rich or poor we are at that time

[16]The equation is called the Ramsey Rule (Azar and Sterner, 1996).
[17]*i.e.* the negative of the income elasticity of marginal utility.

(Bickel and Friedrich, 2005; Clarkson and Deyes, 2002). Azar and Sterner (1996) add to this that in most studies, the rate of discount is assumed constant over time and that this is only correct when we assume that growth in per capita consumption and income elasticity of marginal utility are also relatively stable over time.

The use of the marginal rate of return on investment is based on the fact that capital is productive, *i.e.* a unit of currency will generate more now than in the future. It is reasonable to use the private rate of consumption as a social discount rate when it is assumed that the government acts on behalf of its citizens in undertaking public projects and tries to determine whether the gainers of a policy would be able to compensate the losers.

A more practical approach is the use of the long term interest rate. The latter approach is recommended by the European Commission for its Impact Assessments (EC, 2009), and has also been recommended by the United States Environmental Protection Agency (EPA, 2000b). This has been 4% and relatively constant since the Industrial Revolution (Mendelsohn, R., personal communication, November 2009).

A first comment on discounting is the fact that it is not possible to discount goods and services that have no economic value. This view is supported by the USEPA. Why can we not just discount energy ratios or other physical outputs? These outputs have to be converted into impacts, and these impacts do not necessarily depend on time preference alone, but also on other aspects, *e.g.* it is proven for CO_2 being a stock pollutant that the impact of it gradually grows every year, only to be reaching a certain level where it will have a huge impact. A second comment is that the discount rate should not be adjusted for risky costs and benefits. It is often argued that the discount rate should be adjusted for this risk, and that a high capital cost should be used for uncertain cash flows. This is just too stringent and random (how high should a high capital cost be?) and will discourage investments in long term projects. Moreover, this is not correct because the discount rate would then represent uncertainty of benefits and costs at the same time with the length of time it takes for these costs and benefits to occur. We should keep both goals separate (EPA, 2000b).

1.3.6 Solutions to externalities

Finding a solution to externalities does not mean that external costs are actually avoided. What society would like to do is maximize the social surplus of the production of a certain good. What society then should do, is internalize the external costs (and benefits) of this good. Doing this, Private Marginal Cost and -Benefit curves are moved, resulting in Social Marginal Cost and -Benefit curves. The socially desirable level of production and consumption can be determined using the principles of a perfect market, it is the point where society's marginal valuation of the good coincides with its social marginal cost, and where there is no longer a dead weight loss (Figure 1-2). The next step is to design a policy that actually corrects the externality, *i.e.* makes consumers and producers move to this optimal point. If economists would design environmental policies, they suggest the maximization of efficiency, *i.e.* maximizing the difference between benefits and costs. This does not necessarily have to contradict with equity[18]. The efficient policy might distribute the revenues so that everybody is better off than in a non-efficient situation (Zylicz, 2000). Therefore, the difficulty lies not in discovering the correct price (or the optimal point), but more so, in developing the institutional policy that uses the information gained from economic analysis (Castle, 1970). An efficient policy fully internalizes the externality. An inefficient policy might either not fully internalize the externality, or might even cause a new externality (Mendelsohn, R., personal communication, March 2011).

1.3.6.1 Property rights as a potential solution

Under particular circumstances, *i.e.* when property rights are clearly defined, when transaction costs are sufficiently small, and when redistribution does not influence marginal values, the efficient outcome will occur through bargaining.

[18]Remind that the most common source of critique towards CBT is that it disregards income distribution or equity.

The bargaining occurs between the agent that causes the damage, and the agent that suffers from the damage. Agents will start bargaining up to the point where the marginal damage function equals the marginal abatement function. This is independent on who has the right to damage or the right to not incur damage. Although the fact that who actually has the property rights does not matter for efficiency purposes, it does matter from an equity or distributional standpoint as to which agent will have to pay off the other. So, if the above mentioned conditions are met, externalities will be internalized via the bargaining system. This is regardless of who is assigned the right in the environment, it occurs automatically and does not require government intervention (Coase, 1960). We explain this mechanism with an example in Figure 1-4. However, in most environmental cases, high transaction costs, *i.e.* the costs that have to be incurred to facilitate the transaction, are the cause of failure of the bargaining process.

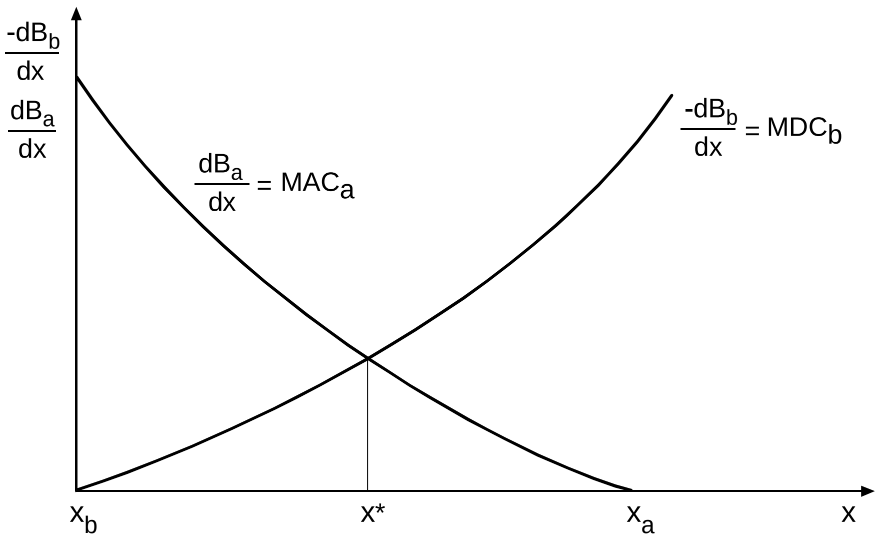


Figure 1-4 Illustration of Coase's theorem with x the intensity of activity a, x* the optimal x, x_a the intensity if a had property rights, x_b the intensity if b had property rights, and MAC_a and MDC_b the marginal benefit function for a and b respectively
Source: based on Lofgren (2000)

x represents the intensity of activity a (Figure 1-4). Let B_a be the net revenue from activity a, given that activity a is conducted at intensity x. Let B_b be the net revenue of activity b when activity a is conducted at intensity x. The socially optimal intensity x of activity a would then be such that the sum of the net revenues from activity a and activity b is maximized. The optimal x* is the x such that the net marginal benefit of activity a equals the net marginal damage of activity a on activity b. x_a would be the level of activity a if activity a was allowed property rights, since x_a is the level where marginal benefits are 0. This is however a private optimum of activity a and too high. The reasoning is the same for activity b. x_b would be the level of activity a if activity b was allowed property rights. Through bargaining by both parties, the economically efficient level of activity a is reached at x*. If activity a has property rights, why would activity b allow a too high level of activity a? Activity b can bribe activity a to have a lower level by paying an amount that lies above his marginal abatement cost function MAC_a (to compensate the foregone earnings), but lies below its own marginal damage cost function MDC_b. Both parties gain from reducing the activity level from x_a to x*. The reasoning is comparable in the other direction. If activity b has property rights, to go from x_b to x*, a will pay b an amount below MAC_a, *i.e.* any amount lower than the foregone earnings will be beneficial to activity a. Activity b will accept any amount that lies above its damage MDC_b.

From this example it is clear that for the Coase theorem to hold it does not matter who has property rights, but it does matter for equity matters, *i.e.* who pays, and who gets paid. If the conditions hold, we always arrive in the social optimum. The magnitude of the benefit though depends on the bargaining process.

1.3.6.2 Command and control (setting standards)

When bargaining is not a solution, due to unmet conditions, such as might be the case when there are many stakeholders involved, direct government regulation, a common approach in the ’60 and ’70, might prove useful. Instead of instituting a legal system of rights, which can be modified by transactions on the market, government may impose regulations which state what people must or must not do. As Barde (2000) puts it: “*The command and control approach consists of the promulgation of laws and regulations prescribing objectives, standards and technologies that polluters must comply with.”* Standards are based on 4 criteria that are environmental, economic, technological, and political. Environmental criteria relate to standards to safeguard health and safety. In this context, uncertainty and irreversibility are important issues to decide what level of risk is acceptable. The technological criterion relates to the BAT (Best Available Technology) principle. Standards are enforceable to the extent that technologies exist or are likely to be developed in the near future. These criteria should be complemented with an economic criterion to make sure that both benefits and costs are evaluated, leading to the BATNEEC principle (BAT Not Entailing Excessive Cost). Finally, political criteria consider issues as equity, simplicity, acceptance, and precaution (Barde, 2000).

The first strength of command and control lies in the fact that there is a lot of experience in the public field. Moreover, sometimes regulation is the correct approach when circumstances require immediate initiative. Lofgren (2000) points out that there are circumstances where direct control is preferred over economic incentives, as is the case for highly toxic substances or urgent matter. This was already indicated by Hahn and Stavins (1992). And finally, regulation standards create a feeling of certainty because once the limit is set, it should not be exceeded, given appropriate enforcement.

However, in the latter immediately lies one of its weaknesses, enforcement is not always possible due to high transaction costs. Moreover, Barde (2000) mentions that standards are subject to bargaining between public authorities and the private sector. Also, standards are static and do not push technological development[19]. Command and control regulations force agents to reach the same goal of pollution control, regardless of their marginal cost of abatement. While *e.g.* maximum emission levels for a specific pollutant such as CO_2 can limit emissions of pollutants with relative ease of compliance monitoring and enforcement, they are often related to high societal costs (Stavins, 1997; EPA, 2000b).

[19] Porter and van der Linde (1995) state however that environmental standards can foster innovation if (i) the approach to innovation is left to the industry and not to the authorities, (ii) regulations do not envision one particular technology, and (iii) uncertainty is eliminated as much as possible to stimulate investment.

1.3.6.3 Performance oriented approach

Rather than mandating a particular technology for compliance, this approach specifies a source's maximum level of pollution and then allows the source to meet this target in whatever manner it chooses. As a consequence, pollution control requirements can be cost-effectively met, and new technology development is not discouraged. Disadvantages of a performance oriented approach are the burden of compliance monitoring and the fact that there is no incentive to reduce emissions beyond the regulated level (EPA, 2000b).

1.3.6.4 Economic based incentives

Although some policy makers would want to claim that the only objective of environmental regulation is protecting the environment, the decision on a correct policy is far more complex because it involves trade-offs between multiple objectives. Policies can therefore be judged based on several criteria. One could choose a policy that is cost-efficient (*i.e.* maximizing V_t)[20] or that is cost-effective (*i.e.* minimizing costs given a fixed goal) (Hahn and Stavins, 1992). Most economists would argue that economic efficiency ought to be one of the fundamental criteria for evaluating regulations on environment.

An (environmental) policy is said to be cost-effective if it achieves a given level of abatement at minimum total cost, if the burden among n sources is divided according to their MAC function (Table 1-3). A policy is efficient when net benefits of abatement are maximized. This means that not only the level of abatement among sources is determined, but also the efficient total level of abatement. Overall efficiency is achieved when the marginal cost of abatement is equal to the marginal damage caused by the pollution for each emitter (Table 1-3) (Tietenberg, 2003; LNE, 2007).

[20]The discussion whether this results in a sustainable solution can be found in 1.3.5.

Table 1-3 Cost-effectiveness and cost-efficiency

Cost-effectiveness criterion	*Cost-efficiency criterion*
Min $\sum_{i=1}^{n} C_i(q_i)$	*Max* $\sum_{i=1}^{n} [B_i(q_i) - C(q_i)]$
s.t. $\sum_{i=1}^{n} q_i \geq Q'$	*s.t.* $\sum_{i=1}^{n} q_i = Q^*$
$C_i(q_i)$= total cost for abatement by source i (i= 1, …, n) of q units q_i= units of *e.g.* pollution abated by the policy by source i Q'= total units of *e.g.* pollution to be abated (= political) to reach e* (the final total emission level)	$B_i(q_i)$= total benefit for abatement by source i (i= 1, …, n) of q units Q*= total units of *e.g.* pollution abated, not necessarily equal to Q'

Efficiency is a difficult concept to apply since environmental costs and benefits are difficult to evaluate in economic terms. This is why the less stringent concept of cost-effectiveness is often used. However, Zylicz (2000) mentions two reasons why the efficiency criterion should be applied in addition. First, even though cost-effectiveness indicates that problems can separately be solved in the cheapest way, the outcome might not be the generally desired solution (amongst all environmental policies). The second reason is that the sum of resources spent on the environment might be too little or too much compared to other sectors.

Command and control regulations on the one hand force sources to reach the same goal of pollution (e*), regardless of the MAC of each of the sources, they are less cost-effective than other approaches and fail to achieve environmental objectives in the least costly manner. Therefore, government could establish a non-uniform standard of pollution to ensure that all sources would pollute up to a level where they face the same MAC. Moreover, these regulations tend to stop the development of new pollution controlling technologies because there just are no economic incentives for agents to exceed the control targets (Stavins, 1997; EPA, 2000b). Compliance with regulations could be achieved through a fine equal to MAC(e*), and corrected for the probability of getting caught in the (in)action.

Economic incentive instruments on the other hand are forms of regulation that encourage behavior through price signals rather than through specific target levels of pollution. Market-based approaches such as taxes and subsidies (Figure 1-5) provide an incentive for firms to achieve a given level of environmental quality at least cost (Hahn and Stavins, 1992). In a tax-system (t), cost-effectiveness is reached because actors will take measures from $t(e_u) > MAC(e_u)$ until MAC(e*) equals t(e*) to avoid the payment of taxes $(t(e_u\text{-}e^*))$. As a consequence, in this point the marginal cost of abatement will be the same for all actors[21]. Cost-efficiency is reached when the tax is set equal to the marginal benefit of emission reduction MDC(e*)[22]. This is tax t*. Agents would still have to pay a tax t* for their emission e* (gray marked zone). If taxes were set too low, all agents would abate up to where they reach the same MAC (equal to t), but this MAC would be lower than the MDC, implying that they would actually be willing to abate further (Figure 1-5, (i)).

The same reasoning can be applied to subsidies. In a subsidy-system (s), cost-effectiveness is reached because actors will take measures from $s(e_u) > MAC(e_u)$, until MAC(e*) = s(e*) and get paid $(s(e_u\text{-}e^*))$[23] (gray marked zone). Cost-efficiency is reached when the subsidy is set equal to the marginal benefit of emission reduction, or the marginal damage cost of emission (e*). This results in subsidy s*. If subsidies were set too high, all agents would abate up to where they reach the same MAC (equal to s), but this MAC would be higher than the MDC, implying that they government is paying subsidies that are higher per unit of emission that the marginal damage of this unit of emission (Figure 1-5, (ii)).

[21]Every agent is now in the same point t(e*), regardless of their MAC function, of the road they had to take to get to this point.
[22]If you put taxes too high, you will ask agents to abate too much, the benefit from abating (MDC) is lower than the tax and a dead weight loss is created, indicating a loss in social welfare. The same is true if taxes are set too low.
[23]This has nothing to do with a change in social welfare whatsoever, it is rather a movement of money amongst economic actors.

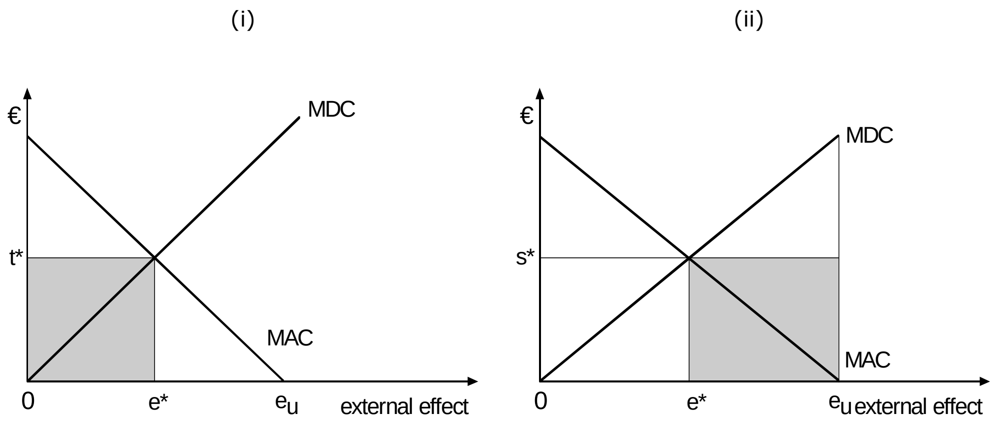


Figure 1-5 Comparison of tax (t) system and subsidy (s) system, with e_u the initial level of external effect and e* the optimal level

Besides the cost-efficiency and cost-effectiveness advantage, taxes and subsidies have another advantage over direct control: they provide an incentive to abate. When a tax or subsidy is imposed, firms are encouraged to develop and apply new technologies (regardless of how much they have already abated), as this will ascertain that they pay less taxes or are paid more subsidies. A disadvantage is still the transaction cost (monitoring).

Valuing policies based on cost-efficiency and cost-effectiveness exhibits two major problems. First, it assumes that all benefits from the policy will be achieved. This might not be the case as an agent might be subject to other important regulations, resulting in transaction costs. Second, in comparing policies, the wrong benchmarks may be used. Therefore, dynamic incentives provided by alternative policies provide a useful criterion, in addition to static efficiency. The effect of policy on technological change might become very important in environmental protection (Hahn and Stavins, 1992; Stavins, 1997).

As opposed to a tax-system, the subsidy-system requires money from the regulator and knowledge of e_u. Lofgren (2000) also mentions that a subsidy would improve the profit conditions for a firm and that, although emissions would decline per agent, this is not necessarily the case on a more general level, as more agents will enter the market due to the subsidies. Indeed, subsidizing renewable energy might not be sending the right signals because it makes energy cheaper. Thornley and Cooper (2008) investigated the effectiveness of government intervention in Sweden, UK, Germany, and Italy, and found that taxation could be an effective means for the development of the bioenergy industry if set at a high enough level to make consumers actually switch fuel, if resulting tax revenues are actually used for investments in the sector, and if applied as a long term measure. In the same countries, overall investment subsidies in renewable energy did not appear to be an effective way of developing the bioenergy industry. Actually, subsidies result in the same cost differentials as taxes, but they are sub-optimal for reasons mentioned above (Eyre, 1997). However, subsidies may be more appropriate than taxes when there are binding income and food consumption constraints (Loehman and Randhir, 1999; Owen, 2006). Subsidies are also easily accepted by the public and often are necessary when the damage has already been done, and the polluter can no longer be taxed (Mendelsohn, R., personal communication, March 2011).

In general (Aidt, 1998), economic agents are motivated to influence environmental policy because the choice of the policy has distributional consequences. If redistribution is the purpose, then a tax system should be chosen over a regulation system (Sunstein, 2005). Firms would prefer regulation over taxes, because the tax imposes an extra burden on the polluter, *i.e.* the tax (gray marked zone), in addition to the abatement cost (represented as the surface under MAC between e_u and e^*)[24]. In case of a standard, the polluter would only pay the abatement cost. Stavins (1998) also indicates this as one of the main reasons why command and control policies have dominated the field for so long.

[24]Again, this has nothing to do with a change in social welfare whatsoever, it is rather a movement of money amongst economic actors.

1.3.6.5 Uncertainty regarding MAC and MDC curves

Since the emitter is paying the tax, or the abatement is subsidized, pollution costs are internalized[25]. So far we have discussed the design of environmental policies under certainty. These kinds of policies are difficult to implement in practice because we need to know the level of pollution at which the MAC and MDC curves cross and it might be hard to determine these marginal functions.

[25]The difference with the regulation system incl. fines is that in a tax system the polluter always pays, in the regulation system, the fine only serves as a threat. This however doesn't matter from a social efficiency point of view, it only has an effect on distributional equity.

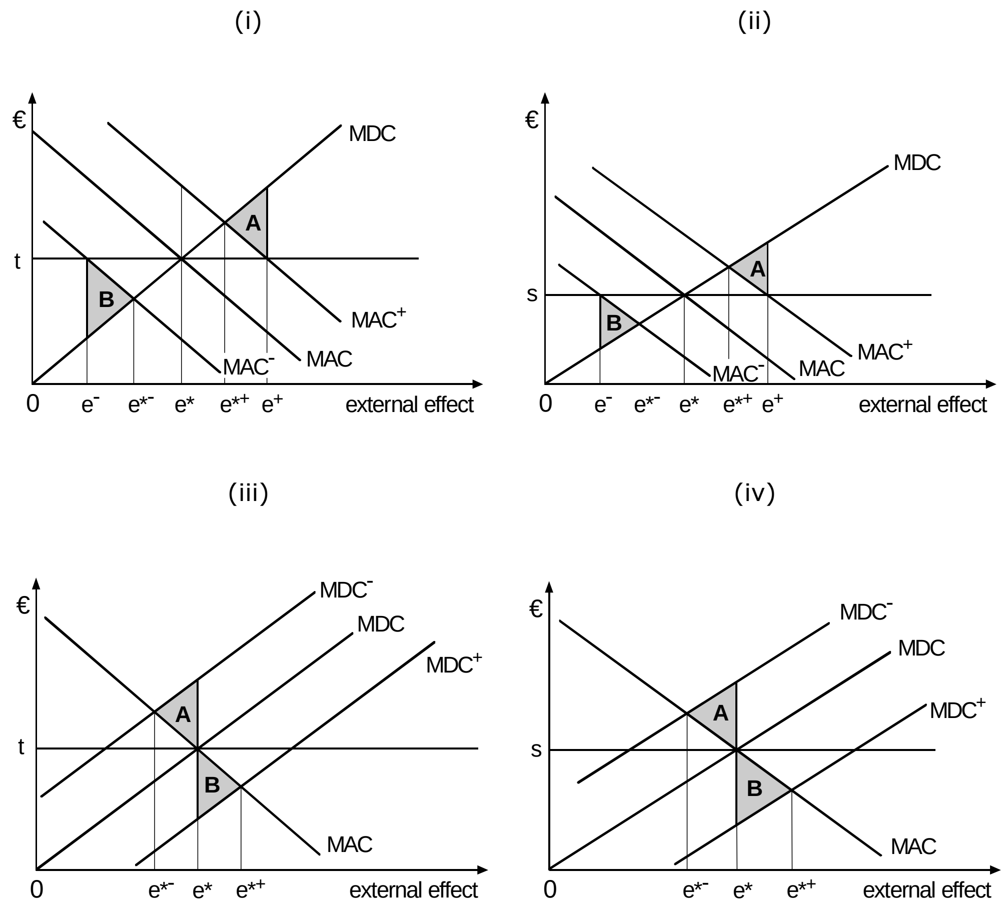


Figure 1-6 Impact in a tax (t) - and subsidy (s) system when the marginal abatement cost function (MAC) is an overestimation (when it is actually MAC^-), or underestimation (when it is actually MAC^+), or when the marginal damage function (MDC) is an overestimation (when it is actually MDC^+), or underestimation (when it is actually MDC^-), with e* the optimal level of external effect for MAC/MDC (e*), MAC^+/MDC^+ (e^{*+}), and MAC^-/MDC^- (e^{*-}), and e^- and e^+ the actual levels of external effect resulting in welfare loss A when underestimating MAC/MDC, and welfare loss B when overestimating MAC/MDC

Suppose we initially assume that MAC is the correct marginal abatement curve, and we base tax t and subsidy s thereon. This results in an optimal level of external effect (e*) (Figure 1-6, (i) and (ii)). However, suppose we underestimated the curve, and that the actual MAC^+ lies to the right of MAC. This means that t and s should have been higher to motivate the reduction in the external effect, and that the optimal level of external effect would have been higher (e^{*+}). Due to the low tax (subsidy) we abate too little, until we reach e^+, the point where MAC^+ equals t. The dead weight loss in underestimating MAC is represented by surface A for both tax- and subsidy system. Suppose we overestimated the curve, and that the actual MAC^- lies to the left of MAC. This means that t and s should have been lower to motivate the reduction in external effect, and that the optimal level of external effect would have been less (e^{*-}). Due to the high tax (subsidy) we abate too much, until we reach e^-, the point where MAC^- equals t. The dead weight loss in overestimating MAC is represented by surface B for both tax- and subsidy system.

Suppose we initially assume that MDC is the correct marginal damage cost curve, and we base tax t and subsidy s thereon. This results in an optimal level of external effect (e^*) (Figure 1-6, (iii) and (iv)). However, suppose we overestimated the curve, and that the actual MDC^+ lies to the right of MDC. This means that t and s should have been lower to motivate the reduction in the external effect, and that the optimal level of external effect would have been lower (e^{*+}). Due to the high tax (subsidy) we abate too much, until we reach e^+ (equal to e^*), the point where MAC equals t. The dead weight loss in overestimating MDC is represented by surface B for both tax- and subsidy system. Suppose we underestimated the curve, and that the actual MDC^- lies to the left of MDC. This means that t and s should have been higher to motivate the reduction in external effect, and that the optimal level of external effect would have been less (e^{*-}). Due to the low tax (subsidy) we abate too little, until we reach e^- (equal to e^*), the point where MAC equals t. The dead weight loss in underestimating MDC is represented by surface A for both tax- and subsidy system.

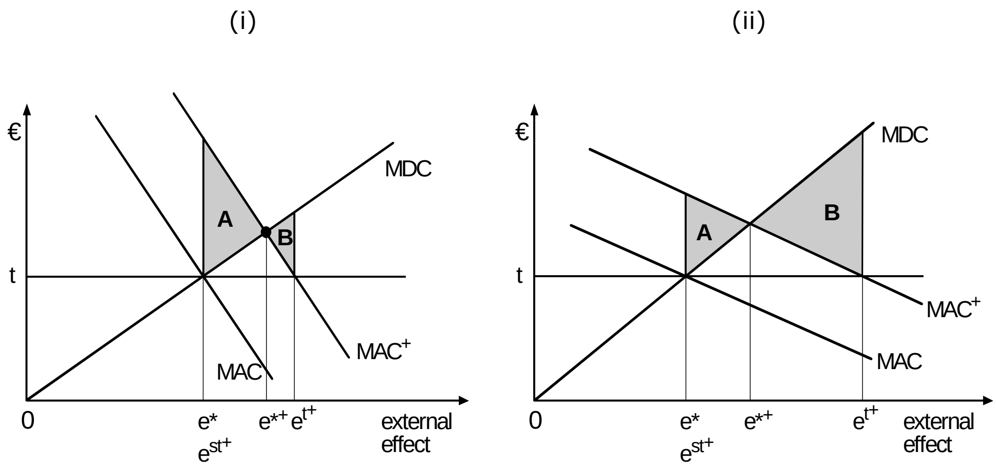


Figure 1-7 Impact of steepness MAC curve ((i) steep, (ii) flat) in a standard system (st) and a tax system (t) when underestimating MAC (when it is actually MAC^+) resulting in welfare loss A in a standard system (st) and welfare loss B in a tax system (t)

The effect of underestimating the MAC curve depends on the steepness of the MAC curve (Figure 1-7). In underestimating the MAC curve, the optimal point of external effect is determined at e*. Given MAC^+, the optimal point of external effect is e^{*+}. A tax system would lead to abatement up until point e^{t+} (where MAC^+ equals t), resulting in a dead weight loss B. A standard system would still lead to abatement up until point e^{st+} equal e*, resulting in a dead weight loss A. In the case of a steep MAC curve (i), standards lead to the greatest dead weight loss (A>B), whereas when the MAC curve is rather flat (ii), standards are preferred over taxes, as the latter lead to a higher dead weight loss (B>A).

1.3.6.6 Should victims be compensated?

Barde (2000) states that damage compensation is efficient in case that damage costs are correctly evaluated, that polluters and victims can be identified, that the causal relationship between pollution and damage can be established, and that the procedure of compensation does not entail excessive costs.

If the number of victims is large, the efficiency criterion tells us that they should not be compensated. The reason can be intuitively related to the fact that taxes are generally preferred over subsidies, as explained. If people would be compensated for the external cost, there would be no incentive for them to not suffer from this external cost (Lofgren, 2000). Moreover, recall the Coase theorem concerning property rights. One of the statements is the fact that the pathway to the optimal point depends on who has the property right. Does the neighborhood have the right on a clean environment and should they be compensated if the environment is polluted? Or does the firm have the right to pollute and should the neighborhood compensate the company to pollute less (Coase, 1960)?

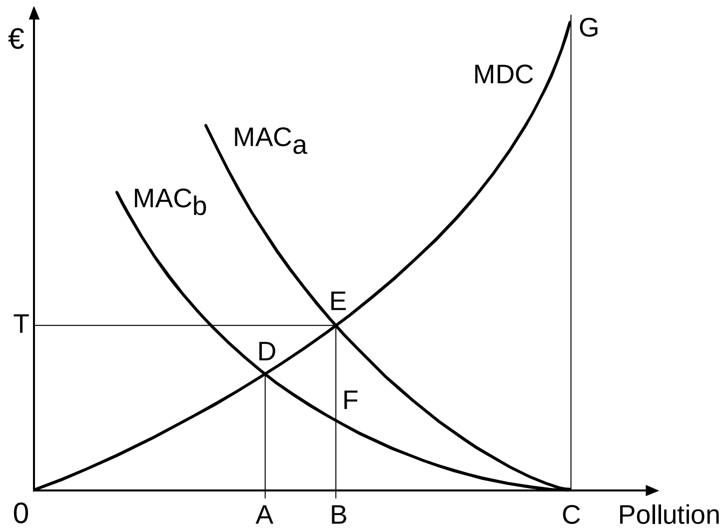


Figure 1-8 Negative externality under different liability rules with MAC_a the marginal abatement cost for the emitter and MAC_b the marginal cost of defensive action
Source: Vatn and Bromley (1997)

Vatn and Bromley (1997) state that given certain conditions (convex curves), when further defensive actions are undertaken by the victims (*i.e.* the recipients of the externality) after abatement by emitters, it might have been cheaper to let these victims carry out more of the remediating action. In Figure 1-8, let MAC_a represent the marginal abatement cost for the emitter and let MAC_b depict the marginal cost of defensive action by the victims. Standard, a Pigouvian tax T would be suggested, leading to a pollution level of B and a net social gain of CEG. After defensive action from the victims, pollution is further reduced from B to A, resulting in an additional gain of DEF. Vatn and Bromley (1997) discuss that if it is economically sound for victims to take defensive action from that point, they might as well have been liable for the whole reduction. Indeed, reducing pollution from C to A, using MAC_b results in an additional social gain of CEF. Vatn and Bromley acknowledge though that MAC_b will most often lie above MAC_a.

1.3.6.7 Critical analysis of externality theory

Vatn and Bromley (1997) state that the model of externalities and foremost the way of internalizing them shows inconsistencies. They state that externalities are caused by transaction costs, which are a function of the internal market (represented by market prices), which leads them to conclude that externalities cannot be analyzed independent from the internal market. This is however the common approach in externality analysis where economists first show the results in a perfect market[26] and then observe that not all goods confirm to this condition, resulting in externalities. This approach does not take into account the interdependency between the internal and external market. The authors are critical about the Pigouvian and Coasean externality theory, where they acknowledge some of the efforts made by the Coasean theory.

First, they argue that the Pareto analysis is not correctly applied to externalities. They state that the Pigouvian solution builds on the conditions for which its solution is aimed, *i.e.* perfectly defined property rights. Even though resources that cause the externality are internalized, the externalities are caused outside the defined property rights and so the mechanisms that apply to a perfect market would not apply to these externalities. Moreover, externalities are often only recognized after they have been caused, requiring not a proactive policy, but a reactive one. To determine the efficient level of resource allocation (*i.e* determination of the optimal/efficient level of externalities), we need to know in advance which party has rights and which one has duties, determined by transaction costs. Vatn and Bromley (1997) then state that the problem with

[26] Recall that in a perfect market there are no information problems, no transaction costs.

externality theory lies in the fact that it makes conclusions about an efficient structure by at the same time relying for its calculations on the existence of this structure.

The second critique is aimed foremost at the Pigouvian analysis where the problem is put too simple that the emitter causes the externality (as opposed to the Coase theorem where externalities are reciprocal, see 1.3.6.1). According to Vatn and Bromley the cause for this problem is arisen because the Pigouvian analysis lacks to acknowledge that the creation of the externality has 2 components: the pure physical one (*e.g.* the emission of CO_2), and the impact of this physical fact (*e.g.* health effect). Both don't have to have the same cause, and the Pigouvian analysis builds on the fact that it does. As a result, they argue that it is not always correct to make the emitter liable, although it might be morally right. Indeed, in economic terms, liability should lie with the party that is best able to make a change with the least cost. Vatn and Bromley (1997) appreciate the attempt of Coase, but critique the fact that he only relies on direct abatement costs and ignores the distribution of transaction costs.

The third critique lies in the rational individual choice on which economic analysis is based to lead to the efficient solution, but at the same time creates transaction costs and thus externalities. To observe externalities as unintended by-products means that they do not fit in the economic system, because the economic system presupposes that all choices are rational and thus intended. Moreover, the authors state that when externalities are defined within an economic system, they will raise in time, resulting in the Pigouvian analysis *"chasing a moving target"*. And finally, the economic system is dynamic, with new technologies leading to a growing economy, inducing externalities. So, the authors ask themselves the question whether to constrain the market expansion ex ante to avoid externalities, whether to tax the externalities generated by the evolving market ex post[27].

[27]This opposes the statement by Aidt (1998) who states that the most efficient instrument to internalize externalities is to aim directly at the source, by which the author means that externalities should be avoided in the first place.

1.3.7 Conclusion

Cost-effectiveness and cost-efficiency are difficult to implement and therefore environmental policies have approached the matter with more practical principles (Zylicz, 2000). According to the polluter pays principle the polluter should pay (a better explanation would be: is forced to take protective measures) whenever legal threshold values/standards are exceeded. This was the principle used in the Flemish Soil Decree (1995). This concept has however been difficult to apply in practice (Section 2). The subsidiarity principle determines policy on the lowest possible level. In the European Union, it finds its application through the use of Directives by which Member States are bound to accomplish an outcome (as opposed to Regulations by which Member States are not only bound to achieve certain goals, but are also bound by the measures they should take to achieve these goals). BAT and more specifically BATNEEC are also often prescribed as the basis to achieve certain threshold values (such as is the case for soil remediation in Flanders, Soil Decree, art 10). This might not be effective (*i.e.* as different technologies imply different MACs), but also not even cost-efficient (as current technologies might just not be enough and polluters are not motivated to develop new ones). The precautionary principle is a strategy to minimize the worst possible outcome. According to the precautionary principle, regulation is required because of scientific uncertainty, even if risks are uncertain. This is the approach that, in our opinion, is now used for food- and fodder threshold values by the European Commission[28] (see 1.4.3). Sunstein (2005) argues that the precautionary principle is incoherent, by stating that risks cannot be eliminated and that by taking actions to avoid risk, other risks will be created.

The policy community seems mesmerized by the possibility of using market-based and incentive-based measures to achieve environmental objectives. Economists tend to search for instruments of public policy that fix problems in one market by introducing another market, while aiming for efficiency in both (Fullerton and Stavins, 1998), such as is the case for biomass production for

[28] Sunstein (2005) states that the precautionary principle is typically used in the European Union, while CBA is used as an organizing principle in the United States.

energy purposes, while guaranteeing that the metal enriched rest product is handled accordingly. However, Hahn and Stavins (1992) are of the opinion that besides efficiency and cost-effectiveness, other criteria should be considered when evaluating governmental policy to reach environmental protection, such as overall effectiveness, ease of implementation, equity (cross-sectionally and intertemporally), information requirements, monitoring and enforcement capability, political feasibility, and clarity to the general public.

The implementation of public policies that satisfy a CBA test does not guarantee sustainability. However, policies that deal with environmental issues at least will improve matters with regards to the sustainability criterion. Little analysis is required to show that an ideal world is better than a state of *laissez faire*. Whatever we may have in mind as our ideal world, it is clear that we have not yet discovered how to get to it from where we are. A better approach seems to be to start our analysis with a situation approximating the one which actually exists, to examine the effects of a proposed policy change, and to attempt to decide whether the new situation would be, in total, better or worse than the original one. In this way, conclusions for policy would have some relevance to the actual situation (Coase, 1960). It is often too easy to find critiques for current policies. Instead of procrastinating remediating action, we should participate in every policy which at least doesn't make the current situation worse, *i.e.* does not reduce economic welfare.

Besides the critiques on policies not resulting in economic efficiency, or lacking a sustainable vision, or focusing solely on economic efficiency and –effectiveness, Rivers and Jaccard (2006) comment that literature on externality analysis uses a too simple perspective of technology as being fixed in time. Therefore they analyze the effect of changing attributes of technologies over time (Figure 1-9). The horizontal curves represent the cost of the conventional technology (where the higher curve represents the cost including external costs caused by the conventional technology), the down-sloped curve represents the cost of the new technology, which goes down in the future due to the learning effect. Even after internalizing the potential external benefits a technology might still be more expensive today than a conventional technology, if we do not take into account the learning effect. Therefore, Rivers and Jaccard (2006) argue that an effort to invest in new, more expensive technology might cause a learning effect and drop the cost (that was not yet taken into account because we have no idea about this learning effect). Therefore, instead of forcing diffusion through regulatory instruments, government should seek to use a strategy based on market instruments. If the total discounted benefit from this technology (B) exceeds the total additional cost of early development of the new technology (A), then the strategy will lead to a net social benefit and should be adopted. What we should find out then is what policy stimulates this type of strategic investment. This might be the focus for further study.

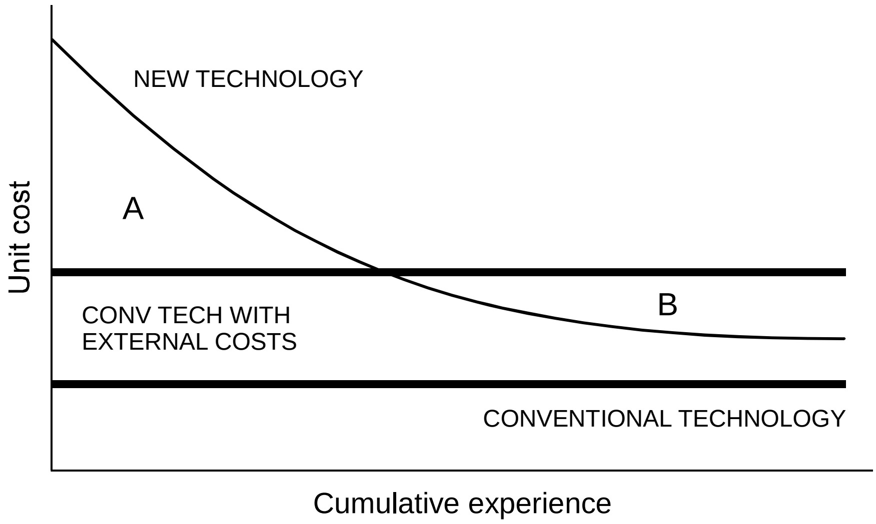


Figure 1-9 Costs and benefits of investments in new technologies with B the total discounted benefit from the new technology and A the total additional cost of early development of the new technology
Source: based on Rivers and Jaccard (2006)

Chapter 1.4 *Legislative issues related to the Campine case*

1.4.1 European Soil Strategy and the new Soil Framework Directive

Research findings and monitoring programs about the status of European soils made the EU decide to analyze and describe the threats being faced by the soils of Europe and to suggest a foundation for their protection. These threats were published in 2002 in a fundamental discussion paper known as "Towards a Thematic Strategy of Soil Protection" (COM(2002)179). The threats considered were erosion, organic matter, contamination, sealing (covering the soil with building materials), compaction, biodiversity, salinization, flooding, and landslides.

In 2006, a Strategy on soil protection was adopted by the European Commission. The purpose of the strategy is to find out what actions and policies would work in the light of both scientific knowledge and past experience (Scape, 2010). The strategy is supposed to function as a mechanism to reach the objectives set out in the 6th Environmental Action Plan adopted by the European Parliament and Council for 2002-2012. Each strategy consists of three phases: a communication, a legislative proposal, and an impact assessment analysing the economic, environmental, and social impacts of the strategy (EC, 2006a). The Soil Thematic Strategy/Communication (COM(2006)231) explains why further action is needed to ensure a high level of soil protection. It sets the overall objective of the Strategy, explains the kind of measures that should be taken, and establishes a ten-year work program for the European Commission. The proposal for a Soil Framework Directive elaborated by the European Commission aims to protect all soils up to a level where their current and future use are guaranteed within reasonable costs, and further contamination and degradation of the soil has to be prevented. All Member States will be obliged to undertake action against different soil threats (contamination being one of them) and will be penalized otherwise.

The current Directive 2004/35/EC (on environmental liability with regards to the prevention of environmental damage and the remedying thereof) did not apply to historical contamination or to damage that was caused prior to its entry into force. As a matter of fact, it did not even cover soil contamination to the same extent as water and biodiversity. Therefore, the new Soil Framework Directive states the following: *"Earlier industrialization and poor or inappropriate management practices have left a legacy of hundreds of thousands of contaminated sites in the Community which call for a common strategy to manage historical contamination of soil in order to prevent and mitigate harmful effects on human health and the environment".* However, in the new Framework, no more details are given on how historical contamination should be dealt with. On 20 December 2007 the EU Council of ministers of Government gathered in Brussels to work on the proposal. Austria, France, Germany, the Netherlands and the United Kingdom did not agree on its content, expectations being high for the next legislature (http://eusoils.jrc.ec.europa.eu/library/jrc_soil/policy). Yet, in 2009, under the presidency of the Czech Republic and Sweden, and in 2010 under the presidency of Spain and Belgium, no improvements have been made yet on the elaboration of the proposal.

1.4.2 Soil remediation threshold values

In Flanders, soil contamination lies within the authority of the Public Waste Agency of Flanders (Dutch: OVAM, Openbare Vlaamse Afvalstoffen Maatschappij)[29]. The decree on soil remediation and soil protection of October 26, 2006 (Soil Decree) is the successor of the Decree of February 22, 1995. The two primary objectives of the Soil Decree are the prevention of new contamination, and the remediation of historical contamination so as to guarantee sustainable soil management. The decree is further executed through a decision by the Flemish Government, the VLAREBO (Flemish regulation on soil remediation and soil protection), and considers soil remediation as historical when caused before October 29, 1995. This is the date on which the first version of the Soil Decree entered into force. After this date, soil contamination is

[29]One of OVAM's missions is the prevention and remediation of soil contamination through obliging the remediation of historical and new soil contamination, developing measures and instruments to stimulate voluntary remediation, developing measures to prevent new contamination, and through the stimulation of brownfield development (www.ovam.be).

considered as new. The contamination in the Campine region is classified as historical.

In the Soil Decree, multiple persons can be denoted as responsible for remediation. The indication of the responsibility follows a gradual system. The first responsible is the exploiter. If this agent does not exist or can prove his innocence, he is exempted from his responsibility and the next agent to be held responsible is the user of the soil. If this agent is exempted, the third agent being held responsible is the owner. If the latter is exempted, the OVAM will act officially. If an agent is held responsible and is not exempted from his duty, he will have to remediate the soil. Costs related to the remediation can however be reclaimed from the agent responsible for the contamination. In 1997, the Flemish Government, Umicore, and the OVAM reached a common agreement (covenant) to deal with historical contamination caused by Umicore. In 2004, the same three actors signed an additional agreement in which it was stated that Umicore would invest € 39,000,000 in the remediation of the adjacent industrial and residential areas (total € 62,000,000), and that it would, together with the Flemish Government, contribute € 15,000,000 to deal with metals in the wider surroundings (total € 30,000,000)[30] (De Turck, 2009).

Concerning new contamination (*i.e.* caused after 1995), the Soil Decree stipulates an obligation to remediate the contaminated soil when soil remediation standards (BSN) are exceeded (art. 9). Such a standard corresponds to a level of soil contamination which entails a considerable risk of harmful effects for man or the environment, taking into account the characteristics of the soil and the functions it fulfills. If there are clear indications that soil contamination (threatens to) exceed BSN, a descriptive soil investigation is carried out immediately. If this investigation shows that BSN have been exceeded, soil remediation is initiated without delay. Remediation

[30]In 2006, samples and questionnaires started in the communities of Balen and Overpelt, followed by the excavation of the soil in the residential areas (www.balen.be; www.overpelt.be). Contaminated soil of the industrial sites in Balen and Overpelt was excavated (80%), and used as cover material. The remediated sites were covered with grass or hard material. In the residential areas (Overpelt-Fabriek, Balen-Wezel and Mol-Wezel) contaminated soil and Zn ashes were removed (www.nyrstar.com).

continues until specified values (target values) for soil quality have been reached. These are determined by the Flemish Government and correspond to a level of contaminating substances or organisms on or in the soil, allowing the soil to fulfill all its functions without the need for imposing any restrictions[31]. If target values cannot be reached because of (i) soil contamination characteristics or (ii) excessively high costs, remediation will occur until BSN are reached. If BSN cannot be reached, for reasons (i) or (ii), remediation will occur up until a level where there is no longer a risk for human kind or the environment. If this is not possible, utilization- and destination restrictions are implied.

For historical contamination, there are no BSN. In accordance with OVAM practices, BSN also are one of the criteria for detecting serious threats in case of historical soil contamination. In order to determine a serious threat, the expert must carry out a risk assessment (OVAM, 2004a). This implies that soil should be remediated from the moment contamination is severe and up until a level where risk for humans or the environment is constrained. If this cannot be reached within reasonable costs or because of soil characteristics, destination and utilization of the soil will need to be restricted (Soil Decree, art. 21). We give an overview of standard soil threshold values in Table 1-4[32].

[31]Background values on the other hand correspond to a level of contaminating matter and organisms in the soil that can be considered as normally occurring in non-contaminated soils with comparable soil characteristics (Soil Decree, Art. 8).
[32]Soil remediation values could also be calculated per parcel, this will not be done in this study.

Table 1-4 STANDARD Background values, target values and BSN for As, Cd, Pb, and Zn

	Background value (mg kg^{-1} dm)	*Target value (mg kg^{-1} dm)*	*BSN according to destination and use (mg kg^{-1} dm)*†				
			BSN I	*BSN II*	*BSN III*	*BSN IV*	*BSN V*
As	16	35	58	58	103	267	267
Cd	0.7	1.2	2	2	6	9.5	30
Pb	31	120	200	200	560	735	1,250
Zn	77	200	333	333	333	1,000	1,250

†BSN I: forestry and nature; BSN II: agriculture; BSN III: residential; BSN IV: recreation; BSN V: industry
Source: VLAREBO appendix II, III, and IV

Standard background values, target values, and BSN can be found in VLAREBO, but should be adjusted for clay and organic material level in the soil (Table 1-5). Based on 100 measure points in the Campine region, the average clay content is determined at 3% and the organic material at 4% (Geysen, D., personal communication, October 2007). For target values and BSN, the pH-KCl value has to be taken into account additionally[33]. Based on 100 measure points, pH is determined at 5 in the Campine region. Soil remediation standards adapted to Campine soil characteristics (pH 5, organic matter 4%, and clay 3%) (mg kg^{-1} dry matter (dm)) are 2, 200, and 282 for Cd, Pb, and Zn respectively.

[33]Site-specific remediation criteria for metals in Flanders have been adapted in 2007 to include soil pH because of the impact of pH on metal mobility and associated environmental risks (Meers *et al.*, 2010).

Table 1-5 ADJUSTED Background values, target values and BSN for Campine soil for As, Cd, Pb, and Zn

	Background value (mg kg^{-1} dm)	*Target value (mg kg^{-1} dm)*	BSN II *(mg kg^{-1} dm)*
As	9†	23†	38†
Cd	0.7	1.2§	2§
Pb	38‡	120	200
Zn	50‡	169¶	282¶

†adjusted for clay content (3%); ‡adjusted for clay (3%) and organic matter (4%); §adjusted for pH (5); ¶adjusted for clay (3%), organic matter (4%) and pH (5)

Recently, there has been a proposal for new BSN (Bierkens *et al.*, 2010). For Cd this would lead to BSN II values of 6.3 mg kg^{-1} (with an exception for celery: BSN II of 0.87 mg kg^{-1}). Adjusted for pH (5), this would result in general BSN II of 3.7 mg kg^{-1}. This has however not (yet) been converted into law. The difference between the new and old BSN lies mainly in new data, a different method of calculation, as explained in Bierkens *et al.* (2010).

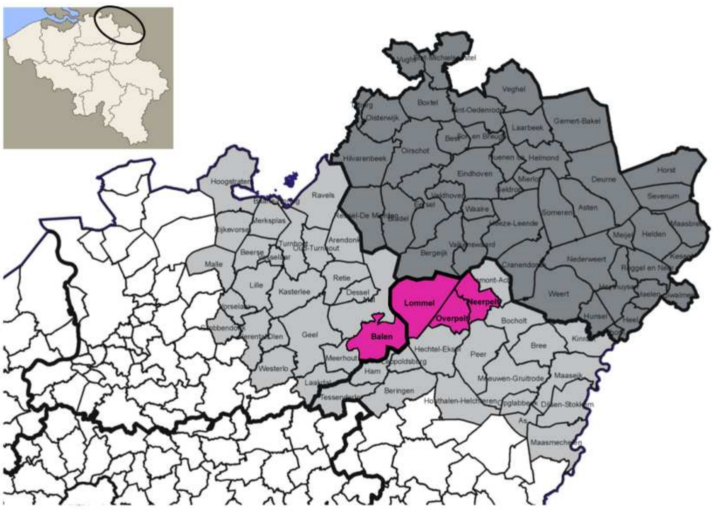


Figure 1-10 Campine region (Belgium + The Netherlands), with an indication of studied area

On the current experimental site (see 3.1.1) BSN II (agricultural use) standards are exceeded for Cd, but not for Pb (although that was close) and Zn (Ruttens *et al.*, 2008). We concentrated our analysis on Balen, Lommel, Overpelt and

Neerpelt. The studied area and its situation in Belgium and the Campine is depicted in Figure 1-10.

1.4.3 European and Belgian product threshold/safety values

Aside from these soil standards, there are European and Belgian product threshold values (safety values). In Europe, Regulation n° 1881/2006 of 19 December 2006 sets maximum levels for certain contaminants in foodstuffs (Table 1-6). This regulation is directly applicable in all Member States of the EU. Threshold values on fodder can be found in Directive 2002/32/EG of 7 May 2002 on undesirable substances in animal feed. This Directive was translated into national legislation via the Fodder Decision (1999), last modified in 2009 (Table 1-7).

Table 1-6 Maximum levels for Cd and Pb for food in Europe and Belgium, expressed as mg^{-1} kg^{-1} fresh matter (fm)† and mg^{-1} kg^{-1} dry matter (dm)‡

	Pb (mg kg^{-1} fm)	*Cd (mg kg^{-1} fm)*	*Cd (mg kg^{-1} dm)*
Raw milk	0.02	0.005	
Meat (cattle, sheep, pigs, poultry)	0.1	0.05	
Liver (cattle, sheep, pigs, poultry)		0.5	
Kidney (cattle, sheep, pigs, poultry)		1	
Cereals	0.2	0.1 (wheat 0.2)	
Legumes	0.2		
Vegetables (excl.*)	0.1	0.05	0.45 (beans); 1(tomatoes); 1.6 (cucumber, asparagus); 0.28 (peas); 0.42 (onion)
*Cabbage	0.3	0.05	0.7
*leaf vegetables	0.3	0.2	3.5 (endive); 4 (lettuce); 2.2 (spinach)
*stem- and root vegetables, potatoes	0.1	0.1	0.9 (carrots); 0.5 (potatoes); 1 (leek, celery); 0.4 (scorzonera)
*celeriac	0.1	0.2	2

†Regulation n° 1881/2006; ‡Smolders *et al*. (2007)

Table 1-7 Maximum levels for Cd and Pb for fodder in Europe (between brackets) and Belgium, expressed as mg kg^{-1} fm† and mg kg^{-1} dm‡

Forage	*Pb mg kg^{-1} 12% water*	*Cd mg kg^{-1} 12% water*	*Cd mg kg^{-1} dm*
Grass (pasture, hay, silage);	30 (40)	1 (1)	1.14 (1.14)
Maize (ear and grain); maize (total)	10 (10)	1 (1)	1.14 (1.14)§

†Fodder Decision; ‡Smolders *et al*. (2007), §for 76% water in maize, this results in 0.27 mg kg^{-1} fm

Food and fodder crops from the region often exceed legal threshold values for Cd (Meers *et al.*, 2010). These crops are either intended to be sold on the market or used as fodder. Intake of Cd occurs via inhalation (lungs) and food (gastro-intestinal tract) (Nawrot *et al.*, 2006; OVAM, 2008b). Environmental exposure to Cd in north-east Belgium, in the vicinity of Zn-Cd smelters, has been associated with renal dysfunction, osteoporosis, a 67% population-attributable risk of lung cancer and other health related problems (Nawrot *et al*., 2008). Due to its high mobility and large effect, even in small doses, Cd seems to be the most acute problem (Garbisu and Alkorta, 2001; Vassilev *et al*., 2004).

We present an overview of activities in the 4 communities in 2007 (Table 1-8). The region is known as a dairy cattle, beef and pig area, and most of the crops are grown for own use on the farm (fodder maize, fodder beets and temporary grassland). Cereals are the second largest activity, with a large contribution of maize. Most of this maize is used on the farm for pigs (roughage) or sold to feed mills. Manufacturing crops in Lommel include rapeseed (12 ha) and sugar beets (28 ha). Potatoes and vegetables in open air (not detailed) represent respectively 3 and 2% of the production area in the selected Campine communities[34].

Table 1-8 Cultivation area in Balen, Lommel, Overpelt and Neerpelt in 2007, expressed in number of hectares and percentage in total

Agricultural activity	*Balen*	*Lommel*	*Overpelt*	*Neerpelt*	*Total*
Cereals (grain)	438.08	298.37	152.58	372.09	1,261.12 (31%)
Manufacturing crops	1.70	40.55	3.00	1.00	46.25 (1%)
Potatoes	12.19	48.64	-	60.97	121.8 (3%)
Fodder +temporary grass	685.30	478.03	574.29	745.03	2,482.65 (61%)
Vegetables in open air	1.80	46.62	6.60	26.74	81.76 (2%)
Set aside land	12.29	20.37	13.81	17.70	64.17 (2%)
Total	*1,155.38*	*932.58*	*750.45*	*1,223.53*	*4,061.94 (100%)*

Source: FOD Economy: SMEs, independent Professions and Energy, personal communication (March 2009)

The primary way vegetables from the region arrive at the consumer is through farmers concluding contracts with the vegetable sector. In 2005, the Federal

[34]Set aside land is abolished since the Health Check of the Common Agricultural Policy (EC, 2008).

Agency for Food Safety (FAVV) observed Cd-concentrations exceeding legal limits in carrots and scorzonera, with a resulting confiscation of the harvests. Apparently, the vegetable sector is no longer willing to conclude contracts in Flemish communities where the risk of surpassing legal threshold values of food is high. This imposes a burden on the agricultural sector, as vegetable growing guarantees a relatively high, reliable and time efficient income. For fodder, the impact for farmers is smaller, but nevertheless disturbing. When crops are used as fodder, the metals are accumulated in kidneys and liver, especially in kidneys values are surpassed (MIRA, 2006; Ruttens *et al.*, 2004). When kidneys are removed (and destroyed), cattle products (meat) are allowed to be sold on the market. Regulation 854/2004 restricts the consumption of liver and kidneys of animals older than 2 years from regions where metals are present in the environment. Based on a Ministerial Decision[35] the export of living cattle (> 18 months) which resided >18 months in Balen, Lommel, Overpelt and Neerpelt is prohibited, and also the human consumption of kidneys of these animals is forbidden (OVAM, 2008a). Export of fodder has not been prohibited yet. In the past, fodder threshold values have been slightly exceeded for grasses, beets and maize. Growing these crops was allowed for own use, but not for the market. Milk production has not yet given any problems (Dries, 2007). Based on data from 2006 from the FAVV, milk, meat and eggs from the Campine region are safe (*i.e.* no elevated metal concentrations) (OVAM, 2008a). All these issues led the Public Waste Agency of Flanders (OVAM) to decide that these soils need proper management, if possible through remediation.

[35]Concerning particular measures for the protection of human health by heavy metal poisoning through food coming from animals.

Section 2

CONCEPTUAL MODEL AND OBJECTIVES

Introduction

A large area (700 km^2 in Belgium and The Netherlands) is moderately contaminated due to atmospheric deposition with cadmium (Cd), Pb, and Zn as the main pollutants (Hogervorst *et al.*, 2007). These metals are mainly concentrated in the upper layer of the soil (0–40 cm) and are spread over a vast area, which in the Belgian part alone covers 280 km^2 (www.ovam.be). Flemish soil standards are exceeded for Cd only (VLAREBO; Ruttens *et al.*, 2008). Moreover, large areas of this contaminated land are currently in agricultural use. The soils in the region are characterized by a sandy texture and relatively low pH (De Temmerman *et al.*, 2003) which entails an enhanced risk for uptake of these metals in crops and leaching to the groundwater, resulting in food- and fodder crops that often exceed European and Belgian legal threshold values for Cd (Directive 2002/32/EG; Commission Regulation 1881/2006; Meers *et al.*, 2010; Ruttens *et al*., 2011). This imposes a serious threat on the profitability of the farming industry and led the Public Waste Agency of Flanders (OVAM) to decide that these soils need proper management, if possible through remediation. Because of the vastness of the contaminated area, it is impossible to apply conventional remediation technologies such as excavation and land filling, biological treatment, physico-chemical treatment (washing), and thermal desorption. These technologies tend to destroy every biological activity in soil and are expensive, with total costs in literature ranging between € 50 and € 560 ton^{-1} soil (McGrath *et al*., 2001; Mulligan *et al.*, 2001). In Flanders, costs range between € 20 and € 60 ton^{-1} soil, depending on the technology (Association for Entrepreneurs in Soil Remediation, Belgium, personal communication, February 2010). Therefore, phytoremediation is suggested as an alternative remediation technology, with costs similar to farming costs. Main barriers to the development of commercially viable phytoextraction procedures for trace elements remain the long time required to remediate soil to standards, and the use/disposal of the contaminated biomass. Based on extraction data from Vangronsveld *et al.* (2009), reducing Cd concentrations in soil from 5 to 2 mg kg^{-1} would take 188 years for energy maize, 361 years for rapeseed, and 120 years for willow. Especially in the case of energy maize and rapeseed, the time needed for decontamination may become less of a constraint if the plant-based technology could be combined with a profit making operation (Robinson *et al.*, 2003;

Vangronsveld *et al*., 2009) and in our case study could actually generate an alternative income for the farmer. Using the obtained biomass of a phytoextraction cycle as an energy resource is therefore attractive (Chaney *et al.*, 1997; Dornburg and Faaij, 2005).

Multiple layer problem

Notwithstanding the dismantlement of the smelter in Lommel in 1974, the conversion from pyrolytic to electrolytic Zn refining in Overpelt in the 1970s, a complete termination of Cd production in Overpelt (1992) and Balen (2002) (Peeters, 2006; Nawrot *et al*., 2008), and regardless the fact that externalities of Cd emissions have been internalized through emission standards, and that emissions now lie within European and Flemish standards, the soil remains polluted with metals, with Cd being the focus of our attention. In 1997, the Flemish Government, Umicore, and the OVAM reached a common agreement (covenant) on how to deal with historical contamination in the region. In 2004, an addendum to this covenant was signed which guarantees that Umicore will remediate the adjacent industrial and residential areas in the coming 15 years. Moreover, Umicore and the Flemish Government agreed on a voluntary financial input to deal with metals in the vicinity in the coming 10 years.

The problem in the Campine is complex, involving multiple stakeholders, but also involving multiple layers. First, it concerns agricultural soil where conventional remediation techniques are inapplicable (see 1.2.1). Therefore, a plant-based technology is suggested as an alternative technology. Second, the damage has been done. This study does not handle avoiding (soil) pollution as an external cost of industrial activities, but rather how best to deal with this (soil) pollution as an existing external cost of industrial activities. The analysis does not consider the internalization of the external cost of emissions as this has already been achieved through regulation through which the industry is obliged to respect standards for emission and immission. Rather, the analysis focuses on the external effects of the resulting soil pollution, *i.e.* the analysis searches to avoid metals present in soil ending up in the food chain. Plant-based technologies seem an economically viable way to deal with the consequences of an inactive policy which for too long left externalities of industrial activities uninternalized. The consequential costs are not only born by the liable agent (industry), but also by farmer and society. Third, the farmer, the owner or tenant of the soil, will bear the costs, if not only mentally by having to switch to other activities on the land, then also economically through potential income losses. Consequences of remediation might not be as positive and clear-cut for him. It is very crucial to understand and define what exactly the benefits might become for the farmer since it is these benefits that will motivate him. This is related to the fourth layer, since legislation on soil contamination of agricultural land faces multiple challenges: (i) the system of control is a trapped system where a first "selection" is based on soil standards, (ii) the basis for legislation is not clear, and (iii) control on application of the legislation is inconsistent. It is within this context that farmers in the Campine region are practicing. And fifth, there are additional externalities resulting from the cleanup and this has consequences for social welfare. What makes the problem at hand more complicated is that the use of plant-based technologies might actually result in externalities[36], which can be internalized through policy. We will see that there already exist policies for these externalities. We will subject them to a thorough analysis and offer an alternative where possible and necessary. This study intends to grasp this multi layer attribute.

When we face an environmental problem as in the Campine region, we should try to approach this in an environmentally responsible way. Sustainable development[37], with sustainable agriculture in particular, has been the subject of numerous conferences and discussions over the last decade. Moreover, it gradually becomes one of the guiding principles in action plans of all kind (Sulser

[36]Today, it is widely recognized that cleanup activities of hazardous waste sites may be the cause of external effects *e.g.* greenhouse gases. In August 2009, the United States Environmental Protection Agency therefore published its proposal called Superfund Green Remediation Strategy which outlines strategic recommendations for cleaner site redevelopment (EPA, 2010).

[37]The concept of sustainability goes back to 1983 when the World Commission on Environment and Development under Gro Harlem Brundtland was gathered by the United Nations to address the growing concern on the accelerating deterioration of human environment and natural resources and their consequences for economic and social development. The basic definition of sustainable development, as defined in their report (1987), is to satisfy the needs of present generations without compromising the needs of future generations.

et al., 2001). Vegter (2001) pointed out in the context of the Clarinet European platform on sustainable contaminated land management (Clarinet, 2002a) that management should include the need for sustainable development besides the choice of an appropriate remediation technology. Also, the steering committee of Soil Conservation and Protection in Europe (SCAPE) has recently published a book where sustainable management of soils in Europe is advocated (Scape, 2010).

In December 2009, the European Common Forum (ECF)[38] and Nicole (Network for Industrially Contaminated Land in Europe) published a position paper on innovative technologies (www.commonforum.eu). In their paper it is argued that cost-effective and sustainable land management technologies are necessary to deal with ever increasing soil management costs. Although the necessary in-situ technologies are already available, their commercial implementation in Europe is low due to a lack of an action plan, which does not only deal with technical barriers but also involves environmental, societal and economic factors. We are of the opinion that the use of plant-based technologies provides the opportunity, not just for the farmer, but also for society, to offer an approach that uses agricultural land in its most efficient way[39].

Sustainability is a vague concept. It is not precise, it cannot be made very precise, and can therefore merely serve as an exact guide for policies that have to do with investments, conservation and resource use. However, it is clear that sustainability is about our obligation to the future. It can be defined as an

[38]The European Common Forum (ECF) on contaminated land started in 1994 as an informal group from national government and agencies in the EU and after organizing Caracas (Concerted Action on Risk Assessment for Contaminated Sites) and Clarinet (Contaminated Land Rehabilitation Network) is now recognized by the European Commission (EC) as an important stakeholder network in the development of a European Union soil protection policy. Clarinet (1998-2001) provided a thematic network on interdisciplinary research by integrating technological, societal and economic aspects for contaminated land management. During regular meetings, common views were developed and expertise was offered to the EC, to relevant stakeholder networks and to EU research projects. The ECF also has position papers on the Soil Framework Directive. ECF members advise their ministers in the EU Soil policy discussion. Belgium (Flanders) is represented by the OVAM (www.commonforum.eu).
[39]Efficiency is here defined in its most widely meaning, and is not restricted to economic efficiency.

obligation to conduct ourselves so that we leave to the future the option or capacity to be as well of as we are, so that we can afford to please ourselves as long as this is not at the expense of future well-being (Solow, 1991).

Sustainability is a basic principle in the proposal for a Soil Framework Directive (see 1.4.1). Its preparatory document (Van Camp *et al.*, 2004) states that soil is essentially considered a non-renewable resource because the degradation rates can be rapid while the formation and regeneration processes are extremely slow. Tahvonen and Kuuluvainen (2000) distinguish between (i) expendable resources, of which the consumption at a certain point in time will not affect the amount of this resource that can be used in the future, (ii) renewable resources of which current use affects future utilization possibilities and for which a rational utilization policy is considered necessary for regeneration, and (iii) depletable resources of which consumption of one unit implies that the stock for the future is reduced forever (regeneration is too far away in the future). In their classification, soil is considered depletable, *i.e.* non-renewable. The main question then is how to allocate between the different generations.

As a matter of fact, environmental and socio-economic problems are often interconnected. One might then argue for a holistic perspective on the problem, because a too narrow focus on one problem at a time can, at worst, make another problem even more serious, or, at best, prevent taking advantage of potential synergy effects (Berndes *et al*., 2008). In our study, phytoremediation combined with biomass production to provide feedstock for the production of various biofuels and bioproducts seems to be a good example of where a holistic perspective is adopted.

Research questions

According to Cho (1971), a characteristic of land that makes it especially vulnerable to externalities is its extension, which makes it inseparable from other parts. Moreover, if this land is then owned and controlled by many different people, transaction costs are that high that, without a policy, externalities will arise (and maintain). We find an example of this in the

Campine region where large areas of land are enriched with metals due to industrial activities, with part of the land in agricultural use.

To avoid metals ending up in the food chain, and thus to avoid the external cost of this soil contamination (*e.g.* health effects), threshold/ safety values for soil, food, and fodder have been established on a European, Belgian, and Flemish level. Cd contents in food and fodder crops grown in the Campine region frequently exceed legal threshold values for food and fodder, resulting in crop confiscation. This imposes a burden on agriculture, and regional policy therefore encourages "proper soil management" of the affected region. One way to ascertain agricultural income and at the same time improve soil quality is by growing alternative non-food crops such as willow (*Salix spp.*) in short rotation coppice (SRC) systems, energy maize (*Zea mays*) and rapeseed (*Brassica napus*). All are to a more or lesser extent extracting metals from the soil, and foremost the harvested biomass can subsequently be used as an input for energy production, resulting in additional CO_2 abatement, but also in rest products with elevated metal concentrations (Figure 2-1).

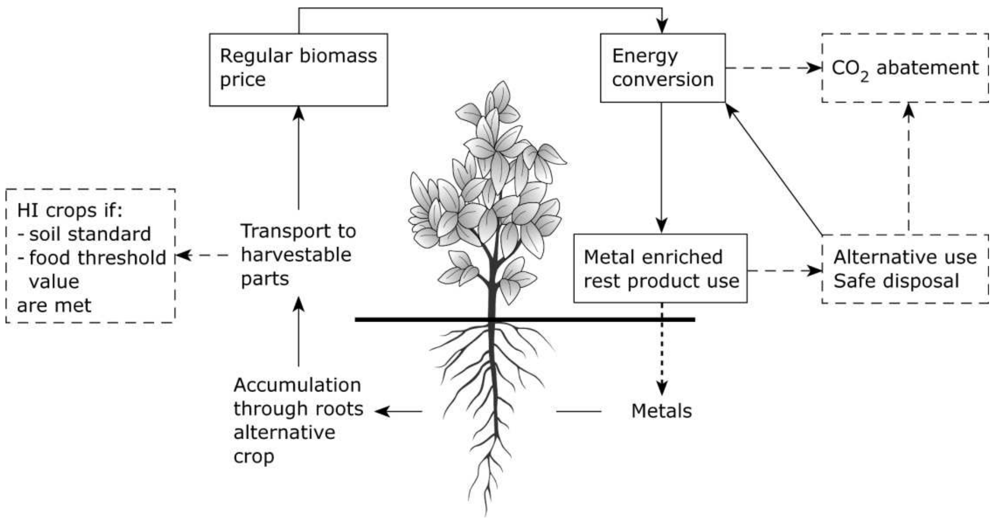


Figure 2-1 Schematic overview of the economic decision model of phytoremediation combined with energy production on farm land

Our economic decision model answers the following research question:

"Does phytoremediation offer a multifunctional and sustainable alternative for conventional remediation technologies for functional repair or management of metal contaminated agricultural sandy soil, resulting in economically optimal remediation strategies using a legislation based business model?"

This rather long sentence includes the following objectives of this study.

- does phytoremediation offer an economically viable alternative for conventional remediation technologies and which crops should be used and under what circumstances? The decisions in this study are based on a cost-benefit analysis.

- does phytoremediation result in other than private costs and benefits and if so, are these externalities positive or negative? The resulting biomass could be used for renewable energy purposes, but also results in a metal enriched rest product[40]. These are considered externalities of phytoremediation and should be internalized through the correct policy.

- do current policies internalize in an economically efficient way the externalities of soil pollution? Do current policies correctly internalize the externalities of phytoremediation?

Given the fact that conventional remediation is not an (economically) manageable alternative (see Chapter 1.2) we suggest phytoremediation as a practical option for the Campine area. We analyse potential remediation crops which do not only serve the purpose of remediation but also (and maybe even more importantly) the purpose of intermediate income generation.

Crops used for remediation of soil pollution need alternative application. We have chosen for energy conversion as a sustainable alternative for several reasons. First, energy production will more likely get public approval, opposed to

[40]We are assuming here that the (elevated) metal concentration in biomass is not an objective property of biomass, and is thus a potential externality (Mendelsohn, R., personal communication, March 2011).

other destinations (*e.g.* paper mills). In Europe, farmers are variously rewarded for direct positive contributions to biological diversity (particularly wildlife habitat), improvements (or avoided negative impacts) to water quality and increased soil health through the concept of cross compliance in the European Common Agricultural Policy (CAP). Many countries also support bioenergy programs, with the intent to promote the production and use of cleaner fuels instead of fossil fuel. Moreover, on a global scale, the increasing interest in carbon sequestering effects of many types of agriculture points to a growing number of programs in the near future that will support certain farming practices as a way of improving overall air quality (DeVries, 2000). Second, energy conversion installations are able to trace metals within their system. At least, as far as we know there is research on this matter in the energy sector, but there has been no research yet on tracing metals in other biomass using technologies (*e.g.* paper mills).

Decision model

The development of an (economic) decision model based on existing legislation involves several steps (Merkhofer, 1987; Hanley, 2000). In a <u>first</u> phase, the quantification of the different benefits of the multifunctional land use requires the identification of physical terms. This involves the determination of (i) current soil characteristics regarding the contamination level. In a next step (ii) different options as to crop choice are defined. This includes energy maize (*Zea mays*), rapeseed (*Brassica napus*), and short rotation coppice (SRC) of willow (*Salix* spp.). Each of these crops will result in different accumulation of metals, different revenues, different energy production, different CO_2 abatement potential and different rest products. In a last step, finally aimed metal levels in soil (iii) are determined, based on food standards for potential high income (HI) crops.

In the BeNeKempen project[41], the cultivation advice is an attempt to indicate at what Cd concentration in soil maximum concentrations in different HI crops are

[41]Soil contamination in the Campine region is that serious and widespread that it has an influence on all aspects of land- and water use in the area. To handle the problem in

respected, given different pH levels in soil. However, the relation between (i) soil pH, (ii) Cd concentration in soil, and (iii) Cd concentration in harvested product is not a one on one relation. For example, at low concentrations of Cd in soil, low as well as high concentrations of Cd are found in crops. Moreover, treatments such as peeling and boiling can cause changes in Cd concentrations in crops. To cover the last problem, there exist well-defined protocols to determine concentrations in harvested crops. To deal with the first issue, the relation between soil pH, Cd concentration in soil and Cd concentration in crops was investigated in more detail. Simultaneous sampling of soil and crops was planned within the BeNeKempen project. Farmers in Flanders were however more reluctant than farmers in the Netherlands. Additional data from earlier measurement campaigns had to be used for the Belgian part. All paired data (2,492) were gathered between 1976 and 2006 and consisted of a simultaneous measurement of Cd in soil and crop, together with a level of pH. Data involved 16 agricultural crops (vegetables), maize and grass. These are referred to as high income (HI) crops as they have the potential to generate a higher income per hectare for the farmer than conventional agricultural crops such as cereals and fodder crops.

In a second phase, physical data are combined in a (business) model by giving them an economic value in a cost-benefit analysis (CBA). Based on the start- and end level of contamination, remediation has different time ranges, depending on the used crop. Total economic value is then calculated as the sum of revenues resulting from the remediating crops, and revenues from crops after remediation, over an infinite time range, based on market prices.

Once the stream of economic costs and benefits is estimated, the standard NPV methodology is applied, using a social discount rate of 4%. As a result, for each initial level of contamination, the crop choice model determines one remediation crop and one HI crop which should be grown successively to maximize the

Belgian and Dutch Limburg, the BeNeKempen project was called into live in 2004, supported by European funding (Interreg IIIa). It involves authorities, stakeholders, administrators, and scientists from both countries.

economic value for farmer and society. Afterwards, we perform a sensitivity analysis, following European guidelines on CBA (EC, 2006b).

The strength of the optimization model lies in its theoretical applicability and comprehensiveness, not in its overall capture of all possibilities: the model reduces the decision for a remediating crop and HI crop to a one-time decision in year 0. We also argue that remediation of soil and sustainable use of soil should not be confused: reapplying rest products (coming from energy conversion of harvested remediation crops) on the original field is considered as a viable option but should be seen in the framework of sustainable soil management and not within the framework of remediation.

Different modeling approaches exist. We use a combination of a partial equilibrium approach and an integrated assessment model. It is a partial equilibrium model since we focus on one sector only without feedback from other sectors[42]. We calculate the effect of environmental policy and of phytoremediation as such on the agricultural sector and do not take into account the effect on *e.g.* the food sector or the energy sector. According to Pollard *et al.* (2004, 2008) successful contaminated land management and policy thereon should be based on interdisciplinary knowledge, combining natural, social and engineering sciences. Moreover, decision making should be considered as a process, based on participation, communication and deliberation. Therefore, in developing the model, we worked closely together with biologists, chemists, and engineers in several projects on metal accumulation in agricultural crops to come to an integrated, multidisciplinary framework.

Challenges

During the analysis, we encounter several challenges. A first challenge is the conventional remediation versus phytoremediation challenge. Phytoremediation

[42]Input Output analysis, introduced by Wassily Leontief, is another modeling technique that describes systematically the different transactions between different sectors in an economy. Macro-economic models explain and predict the economy over time and are driven by changes in the national account. General equilibrium models study the changes in prices, based on the interaction between demand and supply curves (Fankhauser and McCoy, 2000).

has not been applied on a commercial scale in Flanders (Belgium) before. As was shown in Chapter 1.2, the conditions in the Campine region are beneficial for this emerging remediation strategy. The second challenge is the development of the integrated model (Section 4). Such model will simultaneously take account of energy perspectives, biomass production, remediation, and waste disposal. The third challenge involves the theory versus practice paradigm. As long as there is no consistent policy on soil remediation, there are no incentives to put theory into practice. This is also discussed in Section 4.

Phytoremediation

Phytoremediation undoubtedly has a high potential to enhance the degradation and/or removal of organic contaminants from soils (Vangronsveld *et al.*, 2009; Weyens *et al.*, 2009b, 2010). However, based on extrapolations of data obtained from pot experiments, enthusiastic promises have been made concerning the possibilities of metal phytoextraction (Salt *et al*., 1995; Rulkens *et al.*, 1998; Susarla *et al*., 2002; Mench *et al.*, 2010). More demonstration projects on metal extraction are required to provide recommendations and convince regulators, decision makers, and the public of the applicability of plant-based technologies for the treatment of soils, brownfields, groundwater, and wastewater contaminated with toxic metals (and organic pollutants as well) (Vangronsveld *et al.*, 2009).

Therefore, an economic decision tool for the assessment of moderately metal enriched soil in the Campine region (Belgium) has been developed by a multidisciplinary research team, involving stakeholders, assuring long-term ecological, ecotoxicological, social, and financial sustainability, based on data from an experimental field (Lommel). Each incidence of pollution is different and successful sustainable management requires the careful integration of all relevant factors, within the limits set by policy, social acceptance, and available finances.

Moreover, the practical implementation of plant-based technologies has been constrained by the expectation that site remediation should be achieved in a time period comparable to conventional civil-engineering cleanup technologies. This is why alternative use and valorization of the produced biomass, rather than considering it as a waste product of soil remediation, may become a prerequisite for field-scale application of phytoextraction as a remediation technology (Vassilev *et al.*, 2004; Meers *et al.*, 2005a, 2006, 2007a; Vangronsveld *et al.*, 2009). Cost recovery, and the appropriateness of including it as a plant selection criterion, is the subject of increasing current research. Especially the valorization of the biomass as a renewable energy source has promising avenues (Robinson *et al.*, 2003; Vassilev *et al.*, 2004; Ghosh and Singh, 2005; Meers *et al.*, 2005a, 2006, 2010; Vangronsveld *et al.*, 2009).

Decontamination of soil is a long-term goal that can be achieved by striving for short-term goals like producing renewable/green energy to keep the income of the farmers at a level comparable to the situation before the start of remediation (Vassilev *et al.*, 2004; Vangronsveld *et al.*, 2009). In this regard, a number of research projects have been initiated in Flanders, based on the cultivation of industrial non-food crops on contaminated land, with a distinct focus on renewable energy crops (Meers *et al.*, 2007a; Van Ginneken *et al.*, 2007; Vangronsveld *et al.*, 2009; Witters *et al.*, 2009).

Correcting for externalities

Market prices do not always correctly reflect a consumer's Willingness to Pay (WTP). An activity may generate impacts that spill over to other economic agents (see Chapter 1.3). These spill-overs can either be negative or positive. In case of metal enriched agricultural soil where agricultural crops are used for both remediation and energy production, these externalities are (+) CO_2 abatement of biomass based energy because the production and use of this energy will avoid the use of fossil fuels, and (-) the elevated presence of metals in the rest product after energy conversion of the harvested biomass. Therefore, in determining the economic performance of the plant-based technology strategy, adjustments need to be made to the private CBA. Different methodologies are available for estimation of the economic value of externalities (see Chapter 1.3). These externalities are then internalized through policy. If policy is correct, *i.e.* if it correctly internalizes costs and benefits from externalities, this will lead to the economically most efficient solution. If policy is incorrect it will not fully internalize the externality, and might even generate a new externality.

Regarding general soil remediation we find literature on externalities (Andreoli and Tellarini, 2000; Janikowski *et al.*, 2000; Lahdelma *et al.*, 2000). The study performed by Van Wezel *et al.* (2007) gives a comprehensive overview of benefits of less contamination in soil, such as positive health effects, improved drinking water quality, increased property values, … However, most literature on the technology of phytoremediation is fundamental, *i.e.* studies the uptake and translocation of metals, and merely mentions external effects (if even named as such) as a focus for further research. Dushenkov *et al.* (1995) state that the commercialization of rhizofiltration will be driven by economics, by technical advantages, by the reduced volume of secondary waste, the possibility of recycling, and the likelihood of regulatory and public acceptance. The United States Environmental Protection Agency (EPA, 2000a) points to the fact that metal accumulating plants will need to be harvested and recycled or disposed according to applicable regulation. This is also confirmed by Sas-Nowosielska *et al.* (2004) and Ghosh and Singh (2005). Garbisu and Alkorta (2001) put forward that harvestable parts should be easily and safely processed (drying, composting, ashing). Dickinson and Pulford (2005) mention risks to the natural food chains, and to the wider environment during crop growth and after the energy conversion process. However, they perceive these risks as either largely insignificant or manageable. Other possible externalities found in literature relate to biodiversity issues, water control, vegetation filters, biological activity of the soil, carbon sequestration, and erosion control (Burger *et al.*, 2004; Licht and Isebrands, 2005). To the best of our knowledge, there is little literature concerning research on (the actual economic valorization of) externalities resulting from biomass production on contaminated land. Salt *et al.* (1998) mention metal recovery as an option. Borjesson (1999b) and Berndes *et al.*

(2004) point to the fact that Cd in harvested willow stems needs to be collected and deposited in a safe manner.

The externalities of phytoremediation studied here are chosen based on their current appeal (green energy) and urgency (safe disposal of metal enriched rest product). Since both CO_2 and metals are stock pollutants, and accumulate over time, environment has little or no absorptive capacity for them. They can create a burden for future generations by passing on a damage cost which persists well after the benefits received from incurring that damage cost have been forgotten (Tietenberg, 2003). Therefore, necessary policy actions need to be taken to internalize them.

The steps we followed in internalizing both externalities are consistent with the DPSI-R framework which is set out by the European Environment Agency (Bickel and Friedrich, 2005). It describes the causal chain from the origin of the problem to its outcome and potential solution. In this framework, <u>d</u>riving forces put <u>p</u>ressure on the environment, resulting in a change of <u>s</u>tate. This change of state has an <u>i</u>mpact on people and the environment, requiring a <u>r</u>esponse of government.

We first defined the plant-based technology, its resulting potential avoidance of CO_2, and its resulting metal enriched rest product after conversion. Metals should be disposed off safely to avoid negative human contact, CO_2 emission reduction is related to combating climate change. When these impacts are monetized, this results in external benefits and costs, which government can then internalize through correct policies. Indeed, valorizing the soil management function through the biomass production function, taking into account the negative externality of metal enriched waste disposal and the positive externality of CO_2 abatement, will only lead to an economically efficient remediating crop and HI crop choice for farmers in the region, if and only if correct regulations are set into place for each phase, *i.e.* regulations which are based on economic incentives.

Soil quality based on legislation

On a perfect market with well defined property rights and competitive markets, producers and consumers maximize their private surpluses. The price system of this perfect market leads parties to make socially efficient choices. Therefore, in a perfect market, government intervention would not improve social welfare. However, inefficient outcomes occur when the market structure is imperfect and when property rights are not properly defined. The difficulty lies in the development of a consistent, and efficient institutional policy.

Consistent legislation?

The term "consistent" refers to the fact that there exist 2 policies relevant to soil contamination. The first policy is the (Flemish) Soil Decree (1995). Soil standards (BSN) in this decree correspond to levels of soil contamination which entail a considerable risk of harmful effects for man or environment, taking into account the characteristics of the soil (clay, organic matter) and the functions (agricultural, industrial, …) it fulfills. For historical contamination (< 1995) the decree does not define BSN. However, it is OVAM's practice to use BSN as a criterion to detect threats in case of historical soil contamination (OVAM, 2004a). Therefore, to make calculations for the Campine soil, BSN are used as a first guideline to cut the region into an "unsafe" (>BSN) and "safe" (≤BSN) area (classification made by authors). The second policy is the (federal) food and fodder policy based on (i) the European Commission Regulation n° 1881/2006 on product threshold values for food and (ii) Directive 2002/32/EG on fodder standards which defines maximum levels (for Cd) in food- and fodder crops, which has been adopted in Belgium through the Fodder Decision.

Both policies are based on a human toxicological risk assessment of contaminants. Soil standards are easier in use (than food threshold values), since they are averaged to one overall level, based on an average consumption pattern of crops grown on the soil.

The first inconsistency is the fact that checking for compliance with food threshold values after soil standards will lead to vegetables being allowed to be grown in regions that comply with soil standards while they actually exceed food threshold values. Put differently, vegetables are not allowed in the unsafe area, but are allowed in the safe area, although in the latter area metals in certain vegetables also exceed food threshold values. This inconsistency is due to the fact that government is acting in a reasonable way (Mendelsohn, R., personal communication, January 2010). Dividing soils in a safe and unsafe area and then performing expensive controls on food threshold values only in the unsafe area results in lower control costs. Another reason for this inconsistency is the fact that some food crops are highly accumulating Cd and this results in food threshold values being translated in maximum theoretical soil concentrations below background values for Cd in soil. This results in an unfair treatment of farmers in the unsafe area.

The second inconsistency points to the fact that farmers in the Campine region are the focus of more thorough control on food- and fodder threshold values than farmers outside the region. Farmers in the Campine region which have had a soil sample that indicates their soil is safe are tested for food threshold values nevertheless. The reason for this is obvious, but leads to inconsistency.

Efficient legislation?

The term “efficient” refers to the fact that soil remediation policy should be based on an economic analysis. In this case, economic valuation of health impact studies should be the basis for both food-, and soil policies. We discuss some issues with current food- and fodder threshold values and soil standards in 4.1.5. The statement that the presented decision model leads to economically efficient results is based on the fact that current policy on food- and fodder norms, on renewable energy production, and on metal waste disposal do correctly internalize externalities from soil contamination and -management. We perform a mainly qualitative analysis of current policies. We analyze whether current policy on soil remediation does correctly reflect economic costs related to contaminated soil. We also make an exercise based on an economic health risk assessment.

Regarding the renewable energy policy, calculations in Chapter 3.4 and Chapter 3.5 analyze whether current policy tends to oversubsidize biomass for renewable energy. An alternative approach based on the "true" price of CO_2 is offered. Concerning policy on metal waste disposal, our analysis stays rather intuitive and qualitative.

Section 3

CROPS FOR CONTAMINATED AGRICULTURAL SOIL MANAGEMENT: ECONOMIC AND POLICY ISSUES

Chapter 3.1 *Private costs and benefits of plant-based technologies*

Parts of this chapter have been published in:

Witters, N., Van Slycken, S., Ruttens, A., Adriaensen, K., Meers, E., Meiresonne, L., Tack, F.M.G., Thewys, T., Laes, E., Vangronsveld, J. (2009) Short Rotation Coppice for phytoremediation of a Cd-contaminated agricultural area: A sustainability assessment. BioEnergy Research 2(3), p. 144-152

3.1.1 Experimental site

3.1.1.1 Background

Cd concentrations in the region range between values below the background value of 0.7 mg kg^{-1} and above 30 mg kg^{-1} Cd (www.ovam.be). pH-KCl values range between 5 and 5.5 (Geysen, D., personal communication, March 2007). Two large scale experimental fields were installed in the region (Lommel) (one in 2004 and one in 2006 on a former maize field) to evaluate the possibilities of cultivation of non-food crops as an economic alternative for farming on these historically contaminated soils. They are located in Flanders, Belgium (51°12'41"N; 5°14'32"E) and are part of a larger complex of field experiments for phytoremediation research (~ 10 ha) (Ruttens *et al.*, 2008) (Figure 3-1).

The first SRC plantation of willow and poplar in Lommel was planted in April 2004 by Hasselt University on a former maize field. On this field, also maize, rapeseed, and tobacco were grown on small plots.

Based on the results and experiences gained at the first experimental field, a second SRC field was set up next to the original one by Hasselt University, Ghent University, and the Research Institute for Nature and Forest (INBO) where, in addition to some new commercial clones of poplar and willow, several clones of the SRC breeding program of INBO were included. Furthermore, also energy maize and rapeseed were sown on a 1 ha area of the second field to gain experience with the use of these energy crops for phytoremediation purposes. All cultivars and clones were investigated for metal balances, metal extraction, and biomass production.

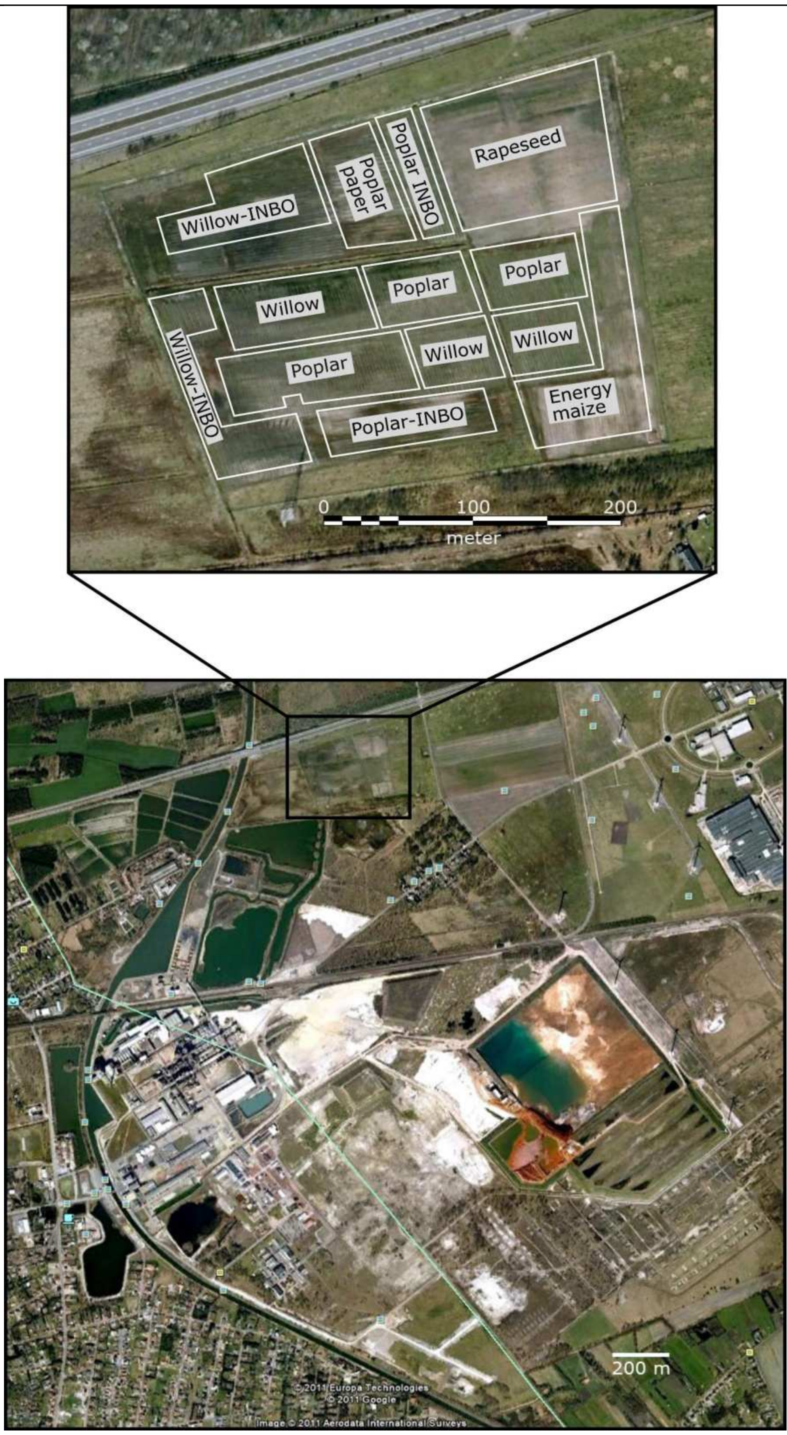


Figure 3-1 Experimental field in Lommel near the former Zinc smelter (Umicore)

Evaluation of these experimental sites offered the opportunities:
- to compare performance of different species, and different cultivars or clones of the same species in specific soil and climatic conditions of the Campine region (biomass production, metal tolerance, and metal accumulation);
- to gain experience with practical aspects of these cultures (weed control, pest control, fertilizer needs, planting, and harvesting techniques);
- to study the effect of growth cycle, rotation length, and planting distance on biomass production of SRC;
- to produce large amounts of biomass to be used in various biomass processing techniques for studying the impact of the presence of metals on the process and its economics;
- to demonstrate the potential of energy crops in the region.

The first plantation mainly dealt with the first two aspects, while the second plantation also focused on the other aspects (Ruttens *et al.*, 2008).

Research for this dissertation was performed in the aftermath of the BeNeKempen project and within the framework of the MIP and CLO project.

<u>BeNeKempen (Demonstration project: Phytoremediation with energy crops)</u>
Soil contamination in the Campine region is that serious and extensive that it has an influence on all aspects of land- and water use in the area. To handle the problem in Belgian and Dutch Limburg and Dutch Noord-Brabant, the BeNeKempen project was called into live in 2004, supported by European funding (Interreg IIIa). It involves authorities, stakeholders, administrators, and scientists from both countries. The project worked on 5 themes: Zn ashes, water, agriculture, nature, and risk assessment. The objective of the project was to develop an integrated attainable management- and remediation strategy for the region and to reduce risks. The project area covers the southeastern part of Noord-Brabant and a small area of Limburg in the Netherlands on the one hand and the northern part of Limburg and the eastern part of Antwerp in Belgium on the other hand. The project aimed at formulating advices on traditional crops and inform the farmer which crops he can grow without running the risk of his

harvest being confiscated. Moreover, the project worked on soil treatment and on the use of water in the region. Besides that, it investigated whether growing alternative crops on the contaminated soils is a valuable farming alternative.

MIP (Remediation of diffusely contaminated soils combined with production of biofuels)

Research partners in this project are (i) VITO (Flemish Institute for Technological Research) for coordination, biodiesel production and energy balances in energy-recovery techniques, (ii) Ghent University for the stimulation of metal uptake by physico-chemical agents, metal balances in soil, and plant and energy-recovery techniques, (iii) Hasselt University for stimulation of metal uptake capacity by the plant-associated microbial community and economic evaluation of phytoremediation combined with (bio)-energy production for remediation of metal polluted sites, and (iv) industrial partners. Industrial partners are Vyncke for incineration in a biomass combustion system, Umicore for incineration in a smelter, EnviTech for gasification, Organic Waste Systems (OWS) for anaerobic digestion, and Indinox for biodiesel production.

CLO (Energy crops on heavy metal enriched agricultural soils)

The research project focusing on the functional restoration by use of phytoextraction and/or biomass production as an alternative for classic agriculture is funded by the Institute for the promotion of Innovation by Science and Technology in Flanders (IWT-Flanders, Grant IWT/CLO/50702). The two strategic goals of this project are (i) the restoration of contaminated soils for conventional agricultural use by means of crops that remediate the soil while generating an income. If this first goal is not possible within a reasonable time period, then (ii) sustainable management of the agricultural soils should be the main purpose with as a main objective generating an alternative income for the farmers.

3.1.1.2 Initial Cd concentration in soil (C_0)

Total soil metal concentrations measured on these fields were in the range of 4.1-7.4 mg kg^{-1} Cd, 160-222 mg kg^{-1} Pb, and 210-418 mg kg^{-1} Zn. Soil pH-KCl ranged between 4.7 and 6.0 (Ruttens *et al.*, 2008). Soil concentrations found by

Van Slycken *et al*. (xxxx) are similar. The concentrations of Cd, Pb, and Zn in the *aqua regia* extracts were respectively 5.3 ± 1.5, 179 ± 36, and 289 ± 15 mg kg^{-1}.

These ranges exceed normal ranges of metal concentration as described by De Temmerman *et al.* (2003). They report that for sandy soils in north-east Flanders concentrations of 0.1-0.5 mg kg^{-1} for Cd, 5-40 mg kg^{-1} for Pb, and 25-70 mg kg^{-1} for Zn are considered normal values. Soil standards (mg kg^{-1}) are 2, 200, and 282 for Cd, Pb, and Zn respectively (VLAREBO). These values are exceeded for Cd, but not for Pb (although that was close), nor Zn.

As a maximum value for the initial level of soil contamination (C_0) we use 12 mg kg^{-1} since this is a common highest measured value in the heavily contaminated zone surrounding the smelters.

3.1.1.3 Concentrations and biomass yield in alternative crops

Energy maize (*Zea mays*) is a crop with low metal uptake capacities (Meers *et al.,* 2005a; Zhang and Banks, 2006), but has the advantage to produce a high biomass yield (leading to a moderate absolute extraction), and the local agricultural sector is very familiar with this crop, cultivated in much the same way as fodder maize. Maize has optimal growing conditions on sandy soils in the Campine at a pH between 5.0 and 6.0, with due attention to good agronomic practices (De Boer *et al*., 2003). Average biomass production of fodder maize lies around 50 ton fm ha^{-1} (Calus *et al*., 2007). In Flanders, it is more likely in the neighbourhood of 45 ton fm ha^{-1}. The presence of Cd can reduce the growth rate of maize (Maksimovic *et al*., 2007). Energy maize crops were sown in May 2007 and harvested in October 2007, with due attention for good agronomic practices. The field was fertilized at a nitrogen dose of approximately 170 kg ha^{-1}. Common agricultural practices were also adopted for land tillage and pest control. This resulted in an average total dry weight of 20±3 ton ha^{-1} (a dry matter percentage of 28-33 %) (Van Slycken *et al.*, xxxx). This translates into a fresh matter yield of 60 ton fm ha^{-1}. This is consistent with biomass yields on non-contaminated soils in the Campine and other Flemish regions (Calus *et al*., 2007; Ghekiere *et al.*, 2008; www.ilvo.vlaanderen.be). Table 3-1 presents an overview of concentrations found in energy maize harvested on the experimental field in Lommel.

Table 3-1 Biomass yield (ton dm) and concentration ranges (min and max) of metals (Cd, Pb, and Zn in mg kg^{-1} dm) in energy maize varieties grown on a metal contaminated field in the Campine (Van Slycken *et al.*, xxxx). Between brackets is the deterministic value used in further calculations (average of 2 varieties)

Organs†	*Yield (ton dm)*	*Cd*	*Pb*	*Zn*
Stem (26 m%)	5.2	0.81-1.68 (1.26)	1.60-3.59 (2.60)	244-398 (337)
Leaves (18.5 m%)	3.7	2.20-3.54 (3.20)	9.80-13.88 (11.34)	443-709 (481)
Bract (6.5 m%)	1.3	0.36-0.88 (0.68)	1.77-3.67 (2.73)	192-314 (256)
Rachis (9 m%)	1.8	0.07-0.62 (0.34)	0.10-1.33 (0.70)	93-223 (149)
Grain (40 m%)	8	0.03-0.35 (0.24)	0.12-0.85 (0.13)	51-65 (58)
Whole plant	20	0.66-1.35 (1.08)	2.40-4.20 (3.06)	186-301 (230)

†m%: this percentage indicates the relative contribution of the various plant parts to the total produced biomass (dm)

Winter rapeseed (*Brassica napus*) is sown half August-September and harvested in July. It can only be grown once every three years. As opposed to maize, the crop is not commonly accepted by farmers, resulting in low agronomic practice (FOD Economy: Statistics, 2006). The technique is well known (Flemish Government, 2005), but not mastered by the average farmer and not straightforward.

In theory, compared to other accumulators that take up metals in rather high concentrations, rapeseed has a high biomass production with a potential biomass yield of 4-6 ton dm ha^{-1} (Grispen *et al*., 2005; Flemish Government, 2005). According to Cidad *et al.* (2003) this lies more around 3.1 ton dm ha^{-1} in Flanders. However, these numbers are not reached in the Campine. The crops sown in 2006 (10 September) and harvested in 2007 (June) had a very low biomass production, mainly due to suboptimal agricultural procedures (fertilization, weed control and pest management). The harvest resulted in an average dry matter yield of green parts of 2.3 ± 0.3 ton ha^{-1} and of seeds of 243 ± 62 kg ha^{-1} (Ruttens, 2008). Calculations are therefore based on a theoretical average dry matter yield of 3 ton ha^{-1} for the seeds and 2.2 ton ha^{-1} for the green parts (0-30 cm and rest) (FOD Economy: Statistics, 2007). Table 3-2 presents an overview of concentrations found in rapeseed harvested on the experimental field in Lommel.

Table 3-2 Biomass yield from literature (ton dm) and concentration ranges of metals (Cd, Pb, and Zn in mg kg^{-1} dm) in rapeseed grown on a metal contaminated field in the Campine (Ruttens, 2008). Between brackets is the deterministic value used in calculations (average of 4 varieties)

Organs†	*Yield (ton dm)*	*Cd*	*Pb*	*Zn*
Seed (57 m%)	3	0.66-1.24 (0.81)	0.07-0.13 (0.12)	67-97 (82)
Green parts (0-30cm)		5.20-6.90 (5.95)	1.63-2.69 (2.12)	354-440 (403)
Green parts (rest)		3.39-5.63 (4.58)	2.32-3.39 (2.77)	265-421 (367)
Green parts (total) (43 m%)	2.2	(5.27)	(2.45)	(385)

†m%: this percentage indicates the relative contribution of the various plant parts to the total produced biomass (dm)

Before 2005, the use of willow (*Salix* spp.) as an income generating crop for farmers was not mentioned in Belgian agricultural statistics (FOD Economy: Statistics, 2006). Recently, experimental plantings have occurred on farm land, but also on the experimental field (Van de Walle *et al.*, 2007a; Ruttens *et al.*, 2008; Meiresonne *et al.*, 2009). The first SRC plantation in Lommel occurred in 2004, a second was installed in 2006. The first harvest of SRC of willow (and poplar) on the experimental field occurred 3 years after planting.

Monitoring the growth of willow revealed that optimal biomass productivity levels were reached after 3 to 4 years. The average biomass productivity was 6 ton dm ha^{-1} $year^{-1}$. This is low in comparison to the average expected productivity values found in literature (10-12 ton dm ha^{-1} $year^{-1}$) (Ceulemans *et* *al.*, 1996; Kopp *et al.*, 2001; Volk *et al.*, 2004). Mean annual shoot yields for two willow clones (Belgisch Rood and Jorunn) in the first rotation cycle were 4.7 and 5.9 ton dm ha^{-1} $year^{-1}$, respectively. We use 4.8 ton (shoot) and 1.2 ton (leaves) in our calculations. This low biomass is not surprising since soil conditions in the area are not very favorable (dry, poor and sandy) for growing these species (Ruttens *et al.*, 2008). It is expected that biomass yields will increase during the next rotation cycles (Witters *et al.*, 2009). Moreover, there were obvious clonal differences: some clones reached levels of 13 ± 5 ton dm ha^{-1} $year^{-1}$.

Metal concentrations in willow shoots were high, resulting in willow having an optimistic phytoextraction potential (Ruttens *et al.*, 2008). Moreover, it is expected that the biomass yield will increase during next rotation cycles. When grown as a source for renewable energy fuel, SRC of willow is usually harvested in winter, to minimize water content in the stem. This means however that leaves have already fallen. This could result in a substantial reduction in phytoextraction effectiveness which could be avoided by harvesting biomass prior to leaf fall, as described by Maxted *et al.* (2007a). Table 3-3 presents an overview of concentrations found in SRC of willow harvested on the experimental field in Lommel.

Table 3-3 Biomass yield (ton dm) and concentration ranges of metals (Cd, Pb, and Zn in mg kg^{-1} dm) in various willow clones grown on a metal contaminated field in the Campine (SRC plantation, shoot results obtained in October of the third year of the first rotation cycle, leaves were analyzed only in the second year) (Ruttens *et al.*, 2008). Between brackets is the deterministic value used in calculations

Organs[†]	*Yield (ton dm)*	*Cd*	*Pb*	*Zn*
Shoot (80 m%)	4.8	15.3-34.5 (25)	14.6-31.1 (22.9)	549-766 (658)
Leaves (20 m%)	1.2	20.4-66.0 (40)	11.3-23.5 (17.4)	2,663-4,249 (3,456)

[†]m%: this percentage indicates the relative contribution of the various plant parts to the total produced biomass (dm)

3.1.1.4 Energy options for biomass from the experimental site

A summary of biomass to energy options used in this study can be found in Table 3-4.

Table 3-4 Biomass conversion options: net energy, reference situation and rest products of biomass coming from plant-based management of contaminated soils,

Option	*Net energy*	*Reference situation*	*Rest product*
Willow			
Co-combustion	Electricity (E)	Natural gas + coal†	Bottom- + fly ashes
Co-combustion	Heat (H)	Cokes‡	Bottom- + fly ashes
Combustion	CHP-steam turbine (E+H)	Separate H+E production	Bottom- + fly ashes
Energy maize			
Digestion	CHP-gas engine (E+H)	Separate H+E production	Digestate
Rapeseed			
Pressing	PPO	Diesel	Cake
Pressing+transesterification	Biodiesel	Diesel	Cake + glycerin

†Electrabel is a large power plant in Belgium. The installation considered in this study lies near the Campine region (Electrabel in Genk-Langerlo); ‡at Nyrstar, a former part of Umicore, a Zinc smelter in the Campine region

3.1.2 *Economic valuation of alternative crops and HI crops*

3.1.2.1 Common Agricultural Policy (CAP)

The European Common Agricultural Policy (CAP) has gone through a lot of changes since its birth in the late ’50. During the first decennia of its existence, farmers were mostly supported with market mechanisms which kept prices at an acceptable level. The Mac Sharry reformation (1992) and Agenda 2000 reformations (2000) both had a large impact on the European support scheme. Traditional price induced income support was replaced in large and replaced by direct support per hectare and per animal (the so-called first CAP pillar). Moreover, obliged set aside land was implied to keep production within control limits. Additionally, Mac Sharry introduced agro-environmental measures while Agenda 2000 introduced rural development as the second pillar of the CAP. In

2003, the Mid Term Review introduced a second big change to the CAP through which subsidies were decoupled even more from production, resulting in yearly single payments (Van Bogaert, 2009). The single payment scheme has been introduced since January 1, 2005 and the value of such a single payment per hectare is based on the historic reference payment on that hectare of land, *i.e.* the average total support the farmer received during 2000-2002 divided by the average yearly number of subsidiable[43] hectares in 2000-2002. Another aid scheme (Mid Term Review) is the energy premium. This is € 45 per hectare of energy crops, with energy crops being defined as crops supplied essentially for the production of biofuels and thermal and electric energy (art. 88) (Regulation 1782/2003).

Equally important for our case is the status of short rotation coppice (SRC). The Health Check (EC, 2008) decided that SRC is eligible for single payments starting from January 2009. The obliged 10% set aside land was permanently abolished through the Health Check (EC, 2008). Also after the Health Check, the European Commission decided in November 2008 that energy premiums do no longer have a function and will no longer be paid (EC, 2008).

[43]Subsidiable hectares include forage area and permanent grass for agricultural activities. In 2005, farmers could not activate payment entitlements on land with permanent crops (art. 43). In 2005, subsidiable crops exclude permanent crops, fruit, vegetables, and potatoes not intended for starch (art. 51). As a consequence of reformations in the sector, vegetables, fruit and potatoes are also eligible as a subsidiable crop from January 2010. A farmer receiving direct payments should respect a number of statutory management requirements referred to as cross-compliance, including (i) minimum standards regarding amongst others the environment, public, animal and plant health, and animal welfare, based on previous European guidelines and regulations, (ii) minimum standards regarding good agricultural and environmental conditions such as soil erosion, organic matter, soil structure and maintenance, and should ensure (iii) that land which was under permanent pasture at the date provided for the area aid applications for 2003 is maintained under permanent pasture (art. 3-5) (Deuninck, 2008).

3.1.2.2 Adapted gross income

Several concepts are of importance when calculating the impact of crop change on the farmer’s income. Common fixed and variable costs in agriculture are represented in Table 3-5, revenues are enumerated in Table 3-6. *Net result* of agricultural production is defined as the money value of the gross production (main product and by-products) minus total costs (variable and fixed) (H-A-B-C-D-E-F-G). *Agricultural income* is defined as total revenues minus total costs, but excluding paid wages (H-B-C-D-E-F-G). *Gross balance* (H-E-F) of agricultural production is defined as the money value of gross production (revenues consisting of main product, by-products, and subsidies) minus related variable costs (excl. taxes). Related variable costs include seed and planting material, fertilizer (purchased), herbicides and pesticides, diverse variable costs (such as irrigation, heat, drying, sorting, cleaning, preparing for sale, insurance, and other specific costs), and animal related costs (such as replacing animals, fodder, roughage, disease protection, production control, and other animal related specific costs). In Flanders, the *gross standard balance* (BSS, bruto standaard saldo) is calculated as the average/standard *gross balance* for each farming category. This balance has long been used by the department of Agriculture and Fisheries as a classification criterion for agriculture. However, due to the decoupling of direct subsidy payments from production (replaced by yearly single payments), the gross standard balance was no longer always positive and could no longer serve as a classification measure (D’Hooghe and Campens, 2009). From 2010, due to Guidelines from the European Union, *Standard Outputs* (SO, standaard opbrengsten) will be used and calculated (Regulation 1242/2008). When calculating these standard outputs, no costs are taken into account, and premiums are omitted.

Table 3-5 Yearly fixed and variable costs on a horticultural and agricultural farm

Fixed costs		*Included costs*
A	Calculated wages	Attributed wage for farmer and family members, based on minimum wages established by the National Joint Committee for agriculture + social security
	Paid wages	Actually paid wages and social security for personnel
B	Buildings and land	Lease†, rent, insurance, maintenance, depreciation
	General costs	Electricity, water, telephone, …
C	Equipment costs	Equipment and machines (depreciation, rent, maintenance, lubricants, insurance)
Variable costs		*Included costs*
D	Third party labor costs	Costs for contractors (when farmer does not have the equipment), such as harrowing, mowing, transport, ...
E	Herbicides, insecticides, pesticides …	Purchased
	Fertilizer	Purchased and/or from own animals, own fertilizer is not valorized as a cost
	Seed and planting material	Purchased
F	Crop costs	Irrigation, heating (electricity, gas, …), drying
	Other crop related costs	Insurance, inspection
	Animal costs	Replacing animals, disposal of manure
	Fodder and roughage	Purchased roughage, fodder from marketable products (milk, hay, …) are conceived as costs, while the use of fodder crops (maize, grass) is not valued as a cost
	Other animal related costs	Insurance, fertilizing, inspection
	Preparation costs	Sorting, cleaning, packaging, converting, …
	Other specific costs	
G	Fuel costs	Fuel, lubricants, …
	Temporary costs	Rent of material, buildings, …
	Selling costs	Auction, promotion, …

Source: Ministerieel Besluit (2007), De Becker *et al.* (2009), Platteau *et al.* (2009), AMS (personal communication, October 2009)

†In general, 50% - 80% of cultivated land per farmer is leased. The lease price is highly dependent on the region (Campine area: € 200-250 ha^{-1}) but is assumed equal for all farmers in the region. The cost is also independent from the activity and is therefore not included in the analysis.

Table 3-6 Yearly total revenues of main- and by-products on a horticultural and agricultural farm

H	Marketable products	Value of agricultural crops: sold, used as fodder or planting material on the farm, or used in household
	Animals	Milk, fertilizer, meat, ...
	Fodder + roughage	See marketable products
	Other revenues	Subsidies, single payments, compensations, ...

Source: De Becker *et al.* (2009), Platteau *et al.* (2009)

The model is based on the adapted gross income (AGI) (€ ha^{-1} $year^{-1}$) (Eq. 6), a method of measurement specific for our purpose. AGI of a high income crop (AGI_{HI}) is based on the gross balance (=H-E-F) because (in Belgium) this is (i) the only measure available on a yearly basis for each crop, and (ii) the only measure that distinguishes between industrial and fresh use of HI crops[44]. In general, revenues from agricultural crops for fresh use (*i.e.* sold locally or at the auction) exceed revenues from the same agricultural crops for industrial use (*i.e.* local food companies such as Noliko[45]).

In accordance with the approach used by the Organization for Economic Cooperation and Development (OECD), the US Department of Agriculture (USDA), and the Food and Agricultural Organization (FAO), and according to general literature on crop rotation modeling, our model does not include wages (A), building costs (B) and selling (auction, promotion) (part of G) costs. However, our approach to add equipment (C), third party labor (D), and fuel costs (G) (based on Flemish data) to H_{HI} differs from (i) international literature on crop modeling, (ii) the approach used by the FAO, and (iii) textbooks. We do add them because SRC of willow and rapeseed are uncommon crops in Belgian agriculture and C, D, and G are important costs to these crops and should thus

[44]We do not use net result (=H-A-B-C-D-E-F-G) because in Belgium (i) data on fixed costs for several HI crops are only available per category of farm, *e.g.* the net result for a farm with vegetables as a head activity but also including other activities, and (ii) costs are not available for different purposes for all HI crops (industrial versus fresh).

[45]For the moment, the vegetable sector (industry) does no longer conclude contracts in Flemish communities where the possibility of surpassing legal threshold values is high. This imposes a burden on the agricultural sector as vegetable growing guarantees a relatively high, reliable and time efficient income.

be added for comparison between energy maize, SRC of willow, rapeseed, and HI crops.

When growing a new crop, the whole farming practice changes as compared to current activities, equipment will have to be bought or rented, or third party labor is needed. Most farmers in the region grow fodder and do not have specialized equipment to grow willow, rapeseed, or vegetables. *Equipment costs* (fixed) are included because SRC of willow and rapeseed need additional large equipment, and the farmer might consider the purchase of additional material. On a yearly basis, these costs can be represented by depreciation costs. *Third party labor costs* (variable) are included when the farmer does not purchase equipment, but rather hires a contractor. *Fuel costs* are also represented in the data on remediating crops (part of the price the farmer will have to pay will also cover the fuel cost of the third party)[46].

The Gross Balance is then adapted for these three costs for comparison between energy maize, willow, rapeseed, and HI crops.

Adapted Gross Income (AGI) (Eq. 6)
= revenue (marketable products (main and rest) + animals + other + subsidies) – cost (third party labor + equipment + herbicides and pesticides + fertilizer + seed and planting material + animal related + crop related + fuel)
= H-C-D-E-F-G

3.1.2.3 Current agricultural activities

Current agricultural activities in the four studied municipalities in the Campine region are summarized in Table 3-7. In the region, dairy cattle farming is the most important activity, with farmers growing fodder maize and temporary grass as feed for the winter period (61%). The other (main) activity is cereals

[46]The reason why the European Commission does not include fuel costs in the Gross Balance (but does include them in the net result) is not clear (AMS, personal communication, October 2009).

(31%). In Lommel, some farmers grow manufacturing crops, potatoes, and vegetables (in open air), while in Neerpelt, some grow potatoes. Idle land is abolished from 2008. These activities do not seem to be the economically most efficient ones, but rather find their incentive in farmers wanting to remain independent for feeding their cattle, convenience, uncertainty regarding price and demand other crops, *etc.* (personal communication with farmers in the region, May 2009).

Table 3-7 Cultivation area (ha) in Balen, Lommel, Overpelt and Neerpelt in 2007

Agricultural activity	*Balen*	*Lommel*	*Overpelt*	*Neerpelt*	*Total*
Cereals (grain)	438.08	298.37	152.58	372.09	1,261.12(31%)
Manufacturing crops	1.70	40.55	3.00	1.00	46.25 (1%)
Potatoes	12.19	48.64	-	60.97	121.8 (3%)
Fodder+temp. grass	685.30	478.03	574.29	745.03	2,482.65 (61%)
Vegetables open air	1.80	46.62	6.60	26.74	81.76 (2%)
Idle land	12.29	20.37	13.81	17.70	64.17 (2%)
Total (ha)	*1,155.4*	*932.58*	*750.45*	*1,223.53*	*4,061.9 (100%)*

Source: FOD Economy: SMEs, independent Professions and Energy, personal communication (October 2009)

Data on costs, biomass yields, and revenues are based on literature. Aegten, a local seed company, added more recent and specific Campine data for third party labor and equipment costs. Aegten assumed that the farmer does not have specialized equipment and will hire contractors. Third party labor costs include fuel costs. Remediating crops are eligible for single payments from the European Union. In 2008, the average single payment[47] in Flanders was € 507. The average single payment for maize is € 450 per ha (Van Broekhoven *et al.*, 2009). Given the background of farmers in the region (Table 3-7), we use this value as a representative average for a Campine farmer.

[47]In 2008, the total amount of single payments added up to € 240,751,481 divided over 474,912 payments (AMS, personal communication, March 2008).

3.1.2.4 Transport

Table 3-8 Transport distances (Dis in km) of biomass and rest products and of ashes after combustion of rest products

Biomass-conversion	*Dis*	*Rest product to second use*	*Dis*	*Ashes to landfill*	*Dis*
Digestion	20	Combustion	30	Landfill cat. 1 or 2	30
		Landfill cat. 1 or 2	30		
		Fertilizer	20		
Biodiesel/PPO	120/0	Fodder	120/0		
		Biogas	30		
		Combustion	30	Landfill cat. 1 or 2	30
		Landfill cat. 1 or 2	30		
		Digestion (glycerin)	30		
(co-)combustion	20	Granulates	30		
		Landfill cat. 1 or 2	30		

We always consider a one-way transport distance (Dis) of 30 km, except for the distance between a biodiesel installation and a farm (120 km), and a digester and a farm (20 km). If digestate is used as fertilizer, the digester has to be located in the immediate vicinity of the farm because transporting digestate with 11 dm% is economically not efficient. When rapeseed is mechanically pressed to PPO, we assume that this is done on the farm, and that the PPO will be used on the farm or in personal vehicles (0 km). When rapeseed is converted into biodiesel, the whole processing takes place at the biodiesel installation. There is thus never a transport of PPO, only of rapeseed (and straw) (Table 3-8).

According to Aegten, transport is included in costs for distances up to 20 km. For distances of 60-70 km, transport costs of € 10 ton^{-1} are on the account of the buyer. For distances of 100-120 km, transport costs of € 20 ton^{-1} are on the account of the buyer. This results in (Eq. 7) (with T = transport cost per ton fresh yield, and Dis = distance). Cidad *et al.* (2003) take account of a transport cost of € 4.2 ton^{-1}, independent of distance. This results in transport costs in Table 3-9.

$$T = (Dis/5) - 4 \qquad \text{(Eq. 7)}$$

Table 3-9 Transport costs (T, € ton^{-1}) for the different phases of the alternative crops: energy maize, rapeseed, and SRC of willow

Biomass-conversion	*T*	*Rest product to second use*	*T*	*Ashes to landfill*	*T*
Digestion	0	Combustion	2	Landfill cat. 1 or 2	2
		Landfill cat. 1 or 2	2		
		Fertilizer	0		
Biodiesel/PPO	20/0	Fodder	20/0		
		Biogas	2		
		Combustion	2	Landfill cat. 1 or 2	2
		Landfill cat. 1 or 2	2		
		Digestion (glycerin)	2		
(co-)combustion	0	Granulates	2		
		Landfill cat. 1 or 2	2		

3.1.2.5 Rapeseed

Table 3-10 PRIVATE variable costs and revenues for rapeseed per ha

Variable costs (€ ha^{-1} y^{-1})		*Variable costs (cont.)*	
Preparation		Stalk control	50
Ploughing	0	Harvest	
Harrowing	0	Seeds	150
Planting		Transport	70
Plant material	60	*Revenues (€ ha^{-1})*	
Planting	65	Seeds	692
Maintenance		Straw	111
Fertilizer	180	Single payment	450
Herbicides	80	Cake for fodder	338
Fert. and herb. application	50	PPO	997-1,588

Source: Suenens (2007), Aegten (personal communication, March 2009)

Table 3-10 gives an overview of agricultural costs and revenues for rapeseed. Rapeseed is only grown once every three years. In the two intermediate years energy maize will be grown. Planting costs include ploughing and harrowing. The average fresh rapeseed yield (20 farmers) in the Campine region is 3,330 kg fm ha^{-1}. Also, 2.22 ton straw is produced. In theory, farmers have several options for the seeds. They can sell rapeseeds as such (to a biodiesel producer), or they can press the seeds and use the oil for personal car use or tractor use[48]. The average of market- and contract prices[49] for rapeseed in 2006 was € 208 ton^{-1}. Straw is sold for € 50 per ton. The distance from the Campine region to the closest biodiesel producer is about 120 km, leading to a transport cost of € 70 per ha (Eq. 7). This is consistent with the transport cost of € 61 per ha of Suenens (2007).

Cold pressing rapeseed results in 1,295 liter PPO ha^{-1} (= 3.33 ton fm $ha^{-1} \cdot$ 350 kg PPO ton^{-1} fm / 0.9 kg PPO $liter^{-1}$ PPO), and 2.16 ton cake ha^{-1} (= 3.33 ton fm $ha^{-1} \cdot$ 650 kg cake ton^{-1} fm). When rapeseed is pressed on the farm, the rest product, cake (2.16 ton) is sold as fodder at € 156.25 ton^{-1}. When rapeseed is sold to a conversion installation, there is no private income from the cake for the farmer. Additional costs for rapeseed pressing (and use on the farm for tractor or personal vehicles) are represented in Table 3-11.

[48]Selling oil to a biodiesel producer is not common. Moreover, this involves a lot of administrative work.

[49]Soy and rapeseed are important sources for the production of oil for food and biodiesel. Prices are therefore world market prices.

Table 3-11 Additional cost on farm (€ ha^{-1} year^{-1}) when rapeseed is pressed to PPO on the farm

Pressing costs		*Rebuilding tractor*		*Rebuilding vehicle*	
Cap. cost press (5 y)	*8,000*	*Cost rebuild engine*	*2,500*	*Cost rebuild engine*	*2,500*
Cap. cost storage tank	500	Deprec. period	10	Deprec. period	10
Cap. cost filter	4,700	Hours tractor y^{-1}	750	Km total period	200,000
Cap. cost press (50 ton y^{-1}, 389 l PPO ton^{-1})	2,800	Oil use (l h^{-1})	5	Oil use (l km^{-1})	0.08
		Total oil use	37,500	Total oil use	16,000
Cost l^{-1} PPO y^{-1}	0.082	Cost l^{-1} PPO y^{-1}	0.067	Cost l^{-1} PPO y^{-1}	0.15625
l PPO ha^{-1}	1,295	l PPO ha^{-1}	1,295	l PPO ha^{-1}	1,295
Pressing cost (€ ha^{-1})	*106.56*	*Rebuilding cost (€ ha^{-1})*	*86.33*	*Rebuilding cost (€ ha^{-1})*	*202.34*

Source: Flemish Government (2005)

3.1.2.6 Energy maize

Table 3-12 PRIVATE variable costs and revenues for energy maize per hectare per year

Variable costs (€ ha^{-1} y^{-1})		*Variable costs (cont.)*	
Preparation		Harvest	
Ploughing	60	Silage	300
Harrowing	40	Transport	0
Planting		*Revenues (€ ha^{-1})*	
Plant material	170	Total plant or grain	1,800
Planting	70	Single payment	450
Maintenance			
Fertilizer	150		
Herbicides	100		
Fert. and herb. application	100		

Source: personal communication with farmers in the region, Aegten, and external firms (2009)

Table 3-12 represents all costs and revenues of energy maize per ha per year. The biomass production of energy maize is estimated at 60 ton fm ha^{-1}. According to local farmers, this could be a maximum because of restricted fertilizer standards (Aegten, personal communication, March 2009). Plant material costs for energy and fodder maize are the same. Before applying fertilizer, farmers apply slurry from cattle or pigs to comply with threshold values from the Manure Decree[50]. Because of the small scale of farms in the region, ploughing, harrowing, planting, and harvesting is done by an external firm. Fertilizer and herbicides are applied by the farmer in 5 hours at € 20 h^{-1}. Silage costs are the same per hectare for energy maize as for fodder maize[51]. When energy maize is sold on the field, the buyer pays harvest and transport (> 20 km), but will pay less for the maize.

[50]In fact, most farmers have land because of this slurry. If the slurry cannot be applied on the land, it will be taken to other land at a cost of € 15 m^{-3}, or transported to a digester or firm to be dried at a cost of € 20-25 m^{-3}. An elaboration of this can be found in Chapter 4.2.

[51]This might change in the future as external firms spend more time on harvesting energy maize than on harvesting fodder maize.

The price of (energy) maize depends on fuel and chemical fertilizer prices. When these prices rise, the price of maize rises too. The price of silage maize depends on the price of maize sold for the grain (Table 3-13): if the price of grains is € 100 ton^{-1}, the price of silage maize is € 30 ton^{-1}, given yields of respectively 15 ton grains and 50 ton silage (fresh matter). When maize is sold on the field, harvest costs are subtracted from the total price, leading to a price per ton fm of € 24 ton^{-1}. Since maize is sold as such to a digester and not converted on the farm, there are no private revenues from the digestate for the farmer.

Table 3-13: Relation between price maize for silage and maize for grain

	Yield (ton fm ha^{-1})	*Price (€ ton^{-1})*	*Revenue (€ ha^{-1})*
Maize (grains)	15	100	1,500
Maize (silage)	50	30	1,500
Maize (on field)	50	24	1,200 (1,500-harvest)

From Table 3-7 one expects that most farmers would very likely feel most comfortable growing energy maize as an alternative crop. This was also confirmed in interviews taken from Campine farmers.

The complete maize harvest is destined for energy conversion in a digester. However, there is a clear difference in metal accumulation by different maize plant parts (Table 3-1). The Cd concentration in the grains is that low that they could safely be used as fodder. The stem, which accumulates more metals, could then be harvested separately and used for energy purposes (digestion). However, there are several issues with this suggestion (unpublished results):

-one would have to harvest the grains in such way that the rest of the plant would be ready for silage;

-the economic efficiency of digestion of maize without grains seems negative due to the high part of grains in the total biogas potential of maize;

-most importantly, the optimal harvest timing is different for grains and stem. In general, when maize is digested in total, the optimal harvest time is at a dry matter percentage of 30%. However, for the grains to be used as fodder, they should be harvested as late as possible. Harvesting at a later time does however change the C/N ratio in the stem and the dry matter percentage (woodification of the stems), which has a negative impact on the biogas potential of the stem. Therefore we conclude that the separate use of grains and stem would be technically, and economically unfeasible.

3.1.2.7 SRC of willow

Table 3-14 PRIVATE variable costs and revenues for SRC of willow per hectare (years indicated between brackets)

Variable costs (€ ha^{-1})		*Variable costs (cont.)*	
Preparation (€ ha^{-1} y^{-1})		End of life cycle (€ ha^{-1})	
Ploughing	67	Stool removal	1,500
Harrowing	75	Harvest and chip	800
Herbicides (3x) + application	240		
Fertilizer + application	90	*Revenues (€ ha^{-1})*	
Plant material	1,800	Stem (per year, every 3 years)	240
Planting	450	Single payment	450
Harvest (€ ha^{-1} every 3 years)			
Harvest and chip	800		
Herbicides	240		
Transport	0		

Source: external firm (personal communication, 2009), and Meiresonne (2006)

At the experimental field, the average yearly harvest of stems and leaves is respectively 4.8 and 1.2 ton dry matter per hectare. Stems have a dm% of 57 (average for 2 willow clones Jorunn and Belgisch Rood for shoot, bark and wood) (unpublished results). Table 3-14 gives an overview of variable costs and revenues for SRC of willow. Most work will be done by an external firm.

An important contribution to the potential introduction of SRC in Flemish agriculture was made in 2006 through the exclusion of SRC from the Forest Decree of 1990 (art. 3 §1). The decree was established in 1990 and puts restrictions on the management of forests in Belgium. Amongst others, art. 20 mentions that in forests it is forbidden to remove plants or parts of it, as well as to use fertilizer and herbicides. SRC is in this decree (art. 4, 14 bis 1) defined as fast growing woody crops from which all the above ground biomass is harvested periodically, maximum 8 years after planting or after the previous harvest, and not planted within vulnerable zones (art. 4). Through the exclusion from the decree, SRC is no longer subject to the strict rules concerning fertilization (although the maximum application has not been determined yet), use of herbicides, and deforestation, … This allows the farmer to efficiently grow SRC without having to fear that his land will be turned into forest land permanently (Decree of 19 May 2006). SRC could also benefit from a change in legislation on farming lease[52]. In current legislation, the farmer cannot plant trees without written consent of the land owner (art. 28) (Law on farming lease)[53].

On the one hand, wood prices depend on wood quality (Vanaken, N., personal communication, February 2009). SRC of willow has characteristics that make it very attractive for (co-)combustion installations: low ash content, low nitrogen content, clean (no iron, sand, stones). This will most likely lead to an interesting price. On the other hand, wood prices depend on the legal status of the wood (Vanaken, N., personal communication, February 2009). When wood is determined as contaminated (C-wood), a fee will need to be paid to "dispose off" the wood. In the other cases (A-wood: untreated wood, and B-wood: uncontaminated treated wood), a positive price will be paid (Table 3-15).

[52]This is national law, so a federal matter.
[53]The farmer can also plant trees on the farm land and can get support according to the Decision of the Flemish Government of 28 March 2003 concerning the subsidizing of forestation of farm lands, implementing Regulation n° 1257/1999 of the Council of 17 May 1999. One of the conditions in the Decision is (art. 8, §3, 8°) that trees can only be harvested after 25 years (for first generation poplar this is 15 years). These trees are no longer seen as SRC and fall within the Forest Decree.

Table 3-15 Price (indications) of different wood types as used by suppliers of wood fuel as of February 2009 (negative prices between brackets)

Type	*Category*	*Price (€ ton^{-1} fm)*	*Price (€ ton^{-1} dm)*
A-wood	Untreated	>20-30	>35-53
B-wood	Treated uncontaminated	25	44
C-wood	Treated contaminated	(40)-(17)	(70)-(30)
Sieve rests	Treated	15-20	26-35
Trim wood	Treated	25	44
Rest of wood processing industry			
Large combustion	Untreated	10-20	18-35
Small combustion	Untreated	20-40	35-70

Source: Vanaken, N. (personal communication, February 2009)

The classification of woody biomass is important for the determination of the appropriate emission directive when (co-) combusted. In Flanders, legislation on emissions is determined in VLAREM I and II. VLAREM I defines the different types of installations and the obligations that follow thereof, while VLAREM II handles the actual emission regulation of each type. We only found maximum concentrations relevant to our case study for Pb (30 mg kg^{-1}). Since Pb concentrations measured in the willow samples from the experimental field lie below these standards, it can be used without any problem. The price for wood is then equal to the price of uncontaminated wood (we use € 50 ton^{-1} dm, or € 28.5 ton^{-1} fm). Finally, prices also depend on the market. Therefore, prices in Table 3-15 should rather be interpreted as how prices between different categories/qualities relate to each other.

3.1.2.8 Overview

All crops receive a single yearly payment of € 450 ha^{-1}. Rapeseed is grown in rotation with energy maize (AGI_{EM} = € 1,260 ha^{-1} $year^{-1}$), resulting in AGI'_{RS} (for rapeseed) as represented in Table 3-16. An average AGI for SRC of willow (AGI_W) is obtained by recalculating the NPV(AGI) over 22 years to an annuity, *i.e.* a yearly constant cash flow (CF) which, after discounting, would again lead to the same NPV. This annuity is obtained by multiplying the NPV(AGI) with the annuity factor (AF) (Eq. 8).

$$AF = i/(1-(1+i)^{-22}) \qquad \text{(Eq. 8)}$$

Table 3-16 AGI, AGI', and r per hectare of alternative crops for different conversion options

	Energy maize	*Rapeseed-PPO-pers use*	*Rapeseed-PPO-tractor*	*Rapeseed-Biodiesel*	*SRC-electr.*	*SRC-heat*	*SRC-CHP*
AGI	1,260.00	1,542.97	1,064.83	548.64	110.72	110.72	110.72
AGI'	1,260.00	1,354.32	1,194.94	1,022.88	110.72	110.72	110.72

AGI = adapted gross income, AGI' takes into account crop rotations

3.1.3 Economic valuation of high income crops

3.1.3.1 AGI_{HI}

Table 3-17 Average (2003-2005) revenue (H), planting costs (E+F), Gross Balance(†), third party labor costs (D)(‡), equipment costs (C)(‡), fuel costs (G)(‡), and AGI for high income (HI) crops, for industrial (Ind) and fresh (Fr) use

HI crop	*Group*	*H*	*E+F*	*Gross Bal*	*C*	*D*	*G*	*AGI*
Potato	(C130000)	3,928	1,196	2,733	68	394	64	2,206
Endive	Ind(C171191)	5,241	3,857	1,384	77	292	76	939
	Fr(C171182)	23,866	7,717	16,149	155	584	151	15,259
Celeriac	Ind(C171061)	3,528	2,027	1,500	77	292	76	1,056
	Fr(C171062)	10,765	3,042	7,723	116	438	114	7,055
Cabbage	Ind(C171291) (C171301) (C171311)	7,256	1,924	5,333	212	0	127	4,993
	Fr(C171292)(C171302) (C171312)	14,605	3,725	10,880	410	0	246	10,224
Oignon	Ind(C171041)	17,250	2,642	14,608	87	254	124	14,143
	Fr(C171042)	14,605	3,725	10,880	123	358	175	10,224
Peas	Ind(C171011)	1,463	441	1,022	35	138	45	804
	Fr(C171012)	21,591	2,559	19,033	203	801	261	17,767
Asparagus	(C171240)	19,411	3,685	15,726	77	292	76	15,281
Beans	Ind(C171021)	1,826	584	1,242	60	104	69	1,009
	Fr(C171022)	14,605	3,725	10,880	383	663	440	9,394
Scorzonera	Ind(C171091)	3,122	1,020	2,102	91	732	84	1,195
Spinach	Ind(C171141)	2,889	1,927	963	56	95	61	750
	Fr(C171142)	15,037	3,231	11,806	94	159	102	11,451
Carrots	Ind(C171031)	2,691	1,345	1,346	41	713	57	535
	Fr(C171032)	15,845	2,974	12,871	91	1,577	126	11,078
Celery	Ind(C171121)	3,564	1,395	2,169	77	292	76	1,724
	Fr(C171122) (C171130)	25,960	4,696	21,264	260	983	255	19,766
Leek	(171250)	14,810	3,758	11,052	488	302	281	9,981
Maize§	(110600)							1,260

†De Becker *et al*. (2009); ‡based on Van Broekhoven *et al.* (2009); §maize is included as an alternative, since this crop is allowed from 10 mg kg^{-1} soil

The gross balance for all HI crops was taken from De Becker *et al.* (2009), using averaged data over 2003-2005 to make price changes endogenous. Regarding equipment costs (C) and third party labor costs (D), existing (international) literature uses one overall cost for all vegetables. In Flanders, differentiated data for HI crops (Van Broekhoven *et al.*, 2009) with data from 2007 exist, and we included them in Table 3-17.

However, data on C, D, and G were not available for all HI crops, so we extrapolated. First, for crops for which we had no data, we used the average of crops for which we did have data. Second, when we had data on C, D, and G for industrial use but not for fresh use, we used the ratio of (E+F) for industrial versus fresh use and applied this same ratio to calculate fresh use for C, D, and G.

The model reduces the decision for the combination of a remediation crop with a HI crop to a one time decision in year 0. While doing this, we should take into account two additional aspects: HI crops cannot be grown year after year and good agricultural practice respects a certain order in growing different crop families. Rotation schemes refer to the combination of different crops in different years.

3.1.3.2 Rotation schemes

First, it should be avoided to plant the same crop (family) year after year. In general this has a positive influence on soil structure, prevents diseases and plagues, and improves quality. In Table 3-18 we give an overview of the HI crops considered in the model, with their family, and minimum years before growing the crop (family) again[54]. Second, not every crop can be grown after every other crop. There is a certain order, which farmers respect for a reason. Potatoes for instance leave behind a loose soil, ideal for fast growing vegetables such as spinach, or root vegetables such as carrots. However, some argue that

[54]Alternatively, one could also rotate HI crops with each other, depending on which one is allowed according to legislation.

carrots after potatoes are susceptible to fungus. After cabbage, which uses a lot of the soil, vegetables such as peas and beans should be grown which enrich the soil with nitrogen again. In the model, HI crops rotate in years as indicated by Table 3-18, and always with energy maize which is allowed whenever HI crops are allowed[55]. Asparagus is grown for 10 years (with 7 harvests) after which the soil needs to recover for 30 years: during the initial 3 years there is no revenue on this hectare. Therefore, we recalculate the income per ha per year to a yearly average, taking into account 7 harvests over 10 years by multiplying the number in Table 3-17 with 7 and dividing by 10. During the 30 years that follow, energy maize is grown.

Table 3-18 Overview of HI crops and rotation scheme

HI crop	*Latin Name*	*Family*	*Rotation*
Potato	*Solanum tuberosum L.*	*Solanaceae*	4
Endive	*Cichorium endivia L.*	*Asteraceae*	3
Celeriac	*Apium graveolens L.* var. rapaceum	*Apium graveolens*	6
Cabbage	*Brassica oleracea*	*Brassicaceae*	6
Oignon	*Allium cepa*	*Alliaceae*	6
Peas	*Pisum sativum L.*	*Leguminosae*	6
Asparagus	*Asparagus officinalis*	*Asparagaceae*	10+30
Beans	*Phaseolus vulgaris L.*	*Leguminosae*	6
Scorzonera	*Scorzonera hispanica L.*	*Asteraceae*	6
Spinach	*Spinacia oleracea L.*	*Amaranthaceae*	3
Carrots	*Daucus carota L.*	*Apium graveolens*	6
Celery	*Apium graveolens L.* var. dulce	*Apium graveolens*	6
Leek	*Allium ampeloprasum L.*	*Alliaceae*	6
Grass	*Lolium perenne*	*Poaceae*	1
Maize (corn)	*Zea mays L.*	*Poaceae*	1
Maize (total)	*Zea mays L.*	*Poaceae*	1
Winter wheat	*Triticum*	*Poaceae*	1

[55]The reason for this simplified approach is to not wander away from the main purpose of this study, *i.e.* offer the farmer a window of opportunities to remediate his land.

For each HI crop we recalculate the average yearly income per hectare to an average over its rotation time, based on (Eq. 9). This results in AGI'_{HI} (Table 3-19).

$$AGI'_{HI}=1/k \cdot [q \cdot AGI_{HI} + (k-q) \cdot AGI_{EM}] \qquad \text{(Eq. 9)}$$

With: k = rotation scheme (Table 3-18)

q = years of consecutive crop growth (10 for asparagus, 1 for all other)

3.1.3.3 AGI'_{HI}

Table 3-19 Adapted gross income (AGI) for alternative high income (HI) crops, corrected for rotation schemes, for industrial (Ind) and fresh (Fr) use, resulting in AGI'$_{HI}$

HI crop¶	*Group*	*AGI$_{HI}$*	*Rotation*	*AGI'$_{HI}$*
Potato	(C130000)	2,206	4	1,497
Endive	Ind(C171191)	939	3	1,153
	Fr(C171182)	15,259	3	5,926
Celeriac	Ind(C171061)	1,056	6	1,226
	Fr(C171062)	7,055	6	2,226
Cabbage	Ind(C171291) (C171301) (C171311)	4,993	6	1,882
	Fr(C171292) (C171302) (C171312)	10,224	6	2,754
Oignon	Ind(C171041)	14,143	6	3,407
	Fr(C171042)	10,224	6	2,754
Peas	Ind(C171011)	804	6	1,184
	Fr(C171012)	17,762	6	4,010
Asparagus	(C171240)	15,281	10+30	3,619
Beans	Ind(C171021)	1,009	6	1,218
	Fr(C171022)	9,394	6	2,616
Scorzonera	Ind(C171091)	1,195	6	1,249
Spinach	Ind(C171141)	750	3	1,090
	Fr(C171142)	11,451	3	4,657
Carrots	Ind(C171031)	535	6	1,139
	Fr(C171032)	11,077	6	2,896
Celery	Ind(C171121)	1,724	6	1,337
	Fr(C171122) (C171130)	19,766	6	4,344
Leek	(C171250)	9,981	6	2,714
Maize	(C110600)	1,260	1	1,260

Chapter 3.2 *Metal enriched biomass for energy*

Parts of this chapter have been submitted in:
Witters, N., Van Slycken, S., Weyens, N., Meers, E., Tack, F., Vangronsveld, J. (xxxx) Does phytoremediation redeem to the expectations of being a sustainable remediation technology: a case study I: Energy production and carbon dioxide abatement. Biomass Bioenerg.

Abstract

In this study, the cultivation of energy crops (energy maize (*Zea mays*), rapeseed (*Brassica napus*), and short rotation coppice of willow (*Salix* spp.)) for combined sustainable valorization and risk minimizing management of metal contaminated soils in Flanders is considered. A crucial aspect when growing energy crops on contaminated soil is the fact whether the harvested crops will be classified as (hazardous) waste, or as biomass since this has an impact on the further utilization and valorization of the crop. Elevated metal concentrations in soils cause increased metal concentrations in plants. Energy conversion of these plants results moreover in a metal enriched rest product, and might therefore lead to substantial risks for human health and the environment in general.

An overview is given of existing legislation on biomass, (biomass) waste and energy crops. More specifically, we unravel the multitude of definitions and regulations that are related to the concept of contaminated energy crops, as this might have technical and thus economic implications for conversion installations. Until now, there is no clear Flemish legislation on this purpose, as there exists no experience with commercial phytoremediation. In Europe neither, as far as we know, but several countries are developing classification schemes for wastes to comply with European Directives (*e.g.* Germany). From the moment we can define the plants as energy crops, consequences on classification are rather clear: we found no evidence of legislation that could exclude energy crops grown on metal enriched land from being classified as biomass. On a European level, the new Waste Framework Directive does not include agricultural waste in its

definition of biomass waste. At the same time, in Flanders, energy crops grown for decontamination purposes could also be considered as biomass waste (and consequently as biomass), but legislation is not conclusive on this aspect. Therefore, we suggest that classification of energy crops with metal accumulating purposes should depend on the main purpose of growing them. If the main purpose is sustainable agricultural land use in function of energy production, crops should be considered as biomass. If the main purpose is remediation, crops could be defined as waste or biomass waste (and in the latter case be considered as biomass) and energy production might be one potential valorization pathway. Phytoremediation would be the main purpose when it involves crops which are capable of accumulating metals such that concentrations in soil decrease substantially (such as with hyperaccumulators). However, there is no quantitative threshold value on this matter. Given the relatively low accumulation potential of the studied crops, we assume that the main purpose is sustainable land management, with energy production for alternative income generation being the main purpose.

For wood waste, the distinction between biomass, biomass waste, and waste might actually become important for conversion, as different legislation will apply depending on classification, having implications on types of combustion installations and emission control. For energy maize and rapeseed, there is no explicit legislation on input for energy conversion installations, but there are threshold values for the output/rest product. For the three studied crops, based on literature, we decided for our case study that metals have no marginal effect on the energy conversion (technical) efficiency. Moreover, no metals end up in the energy carrier, but are concentrated in the rest products: ashes, digestate, and cake. The concentration of metals in the input determines the concentration of metals in the rest product and the latter has to comply with threshold values. We studied the marginal impact of the contamination on processing the rest products for secondary use or disposal, since a different end use might have to be found for the rest products due to the contamination.

3.2.1 Introduction

Elevated metal concentrations in soils may result in increased metal concentrations in plants and therefore in a substantial health risk to man (Vangronsveld *et al.*, 1995). In the following parts we analysed the answer of policy makers to this matter, and in case there is no specific legislation yet, we analysed how existing legislation could be applied to the specific case of energy crops grown on metal contaminated soil.

We should not consider all harvested crops resulting from risk minimizing land management as equal. First, we should make a distinction between harvested crops based on the main purpose of the crops. Second, crops used in this study do not exhibit a high metal accumulating potential and might therefore not be substantially different from "regular" biomass. Therefore, we should put things in their perspective when classifying crops used on metal contaminated land, and use common sense. That's why we will *e.g.* compare the combustion of willow grown on contaminated land with the combustion of coal. Third, if crops are destined for energy conversion, we should analyze the rest product after energy conversion for its contaminant levels. The output might (in some cases, such as with non-hyperaccumulators, very likely) comply with threshold values regarding metals for secondary use of rest products from energy production.

There will always be the need to dispose off residues in the environment because some toxic constituents cannot be destroyed, or are not commercially interesting. Most literature on the technology of phytoremediation is fundamental, *i.e.* studies the process of metal extraction and -translocation in itself, and merely mentions the external effects (if even named as such), There is little literature concerning research on (the actual economic valorization of) externalities resulting from biomass production on contaminated land.

"*The lack of understanding pertaining to metal uptake and translocation mechanisms, enhancement amendments and external effects of phytoremediation is hindering its full scale implementation*" (Alkorta *et al.*, 2004).

3.2.2 Classification of biomass from plant-based technologies and used for energy production

A new aspect regarding the use of energy crops for combined remediation of soils and energy purposes is the fact whether the harvested metal enriched energy crops will be classified as (hazardous) waste, biomass waste, or as biomass.

In the following section, we give an overview of existing legislation on biomass, biomass waste, and energy crops. More specifically, we unravel the multitude of definitions and regulations that are related to the concept of contaminated energy crops since this might have technical and economic implications for conversion installations. Until now, due to a lack of experience in Flanders concerning energy crops for phytoremediation, there exists no clear legislation on this matter (Geysen, D., personal communication, January 2008). To the best of our knowledge, in Europe neither, but several countries are developing classification schemes for wastes to comply with European Directives.

Waste is in the Waste Directive (2006), and its successor, the Waste Framework (WF) Directive (2008), defined as *"any substance or object in the categories set out in Annex I which the holder discards or intends or is required to discard"*. Annex I contains 16 categories of waste. The definition has been taken over in the Flemish Waste Decree, while the annex has been taken over in VLAREA, appendix 1.2.1 A. Additionally, Art. 1.2.1 of VLAREA states that for a substance to be classified as waste it should also occur in a list of waste (Appendix 1.2.1. B). All three conditions (definition, fall within category, and mentioned in list) have to be fulfilled for a substance to be defined as waste. However, classification is not always a clear-cut decision and that's when interpretation of the OVAM is needed. In case it is decided that a substance is waste, it can even be classified as hazardous waste[56] if it exposes traits such as described in Art. 2.4.1 of VLAREA.

---

[56]Hazardous wastes are wastes that (potentially) pose a particular threat for human health or the environment in general or that should be handled in specialized installations. The Flemish Government decides which wastes are hazardous according to European

In Flanders, the Decision of March 5, 2004 concerning the promotion of electricity generation from renewable energy sources defines biomass as *"the biodegradable fraction of products, waste and residues from biological origin, from agriculture (incl. vegetal and animal substances), from forestry and related industries including fisheries and aquaculture, as well as the biodegradable fraction of industrial and municipal waste"*. This definition (with no specific category for biomass waste) is based on European Directive 2001/77/EG[57] concerning the promotion of electricity generation from renewable energy sources on the internal electricity market. This general interpretation of biomass (*i.e.* also including biomass waste in the biomass category) might actually take pressure off the "pure" biomass market for energy production.

Similarly, Directive 2000/76/EG on the combustion of waste[58] (WI) and Directive 2001/80/EG on large combustion installations (LCP) also consider biomass waste under the common denominator of biomass, without the creation of a separate category for biomass waste[59]. The Flemish Regulation concerning the Environmental Permit (VLAREM) II[60] then bases its definition of biomass on these combustion directives. Biomass is conceived as products consisting of any whole or part of a vegetable matter from agriculture or forestry which can be used as fuel for the purpose of recovering its energy. Biomass waste is more specifically considered as (i) vegetable waste coming from agriculture and forestry, (ii) vegetable waste coming from the food processing industry, (iii)

guidelines (Waste Decree). Alternatively, hazardous wastes can be defined as *"a subset of all solid and liquid wastes, which are disposed of on land rather than being shunted directly into the air or water, and which have the potential to adversely affect human health and the environment"* (Dower, 1990).

[57]Directive 2001/77/EC shall be repealed with effect from 1 January 2012 and be replaced by Directive 2009/28/EC.

[58]WI integrates former directives on the incineration of hazardous waste (Directive 94/67/EC) and household waste (Directives 89/369/EEC and 89/429/EEC).

[59]However, as opposed to Directive 2001/77/EC, some waste (such as rest products and animal waste) and manure are considered separately, and are not considered biomass.

[60]VLAREM I and II are both regulations that determine the requirements of the Decree of 28 June 1985 concerning the Environmental Permit more specifically. VLAREM I categorizes the different installations, and provides information on procedures and principles, whereas VLAREM II defines general and sectoral emission levels, based on the classification in VLAREM I.

fibrous vegetable waste coming from pulp from the paper industry, (iv) non treated wood waste, and (v) cork waste.

Also, in the Biomass Inventory 2007-2008, biomass includes (i) "biomass waste", *i.e.* the (separated) gathered biological degradable fraction of industrial- and household waste and (ii) "other biomass" including biomass from the agricultural sector such as vegetal products, and energy crops defined as crops grown with the intention to use for energy production (*e.g.* SRC and grain) as far as there is information available at the involved competent authorities (OVAM, 2010).

Based on the above, we can only conclude that in general, energy crops are considered as "other biomass", and that the wide definition of biomass also includes "biomass waste", in European legislation implicitly, in Flemish legislation explicitly.

The next step is to find out whether metal contaminated energy crops fall within the definition of energy crops (and thus are "other biomass" and thus biomass) and if not, whether they fall within the definition of "biomass waste" and as a consequence are also considered biomass (as opposed to "waste"). As a general remark, the term "contaminated" has to be put into perspective and has to be interpreted as "enriched", based on rather low concentrations in plant parts of the studied crops (Table 3-1, Table 3-2, Table 3-3).

Neither in the "action plan for wood waste", nor in the "action plan for organic biological waste" energy crops are mentioned. According to the Public Waste Agency of Flanders (OVAM) energy crops are grown with the purpose of energy generation and as such do not need to be considered as biomass waste. Therefore, energy crops fall outside OVAM's competence (OVAM, 2004b; 2006). Thus, since the OVAM has no experience with energy crops considered as biomass waste, we cannot draw any conclusions on whether or not contaminated energy crops are considered as energy crops (other biomass).

On a European level, the WF Directive defines biomass waste as "*biodegradable garden and park waste, food and kitchen waste from households, restaurants, caterers and retail premises and comparable waste from food processing plants.*" This is an addition to the Waste Directive. In this definition, we see no opportunity to include energy crops for phytoremediation within the category of biomass waste. Also, in the Biomass Inventory, we see no opportunity for energy crops to be included in the definition of biomass waste. On Flemish level, VLAREM II does include vegetable waste coming from agriculture and mentions a distinction for wood (but not for energy maize or rapeseed). "Contaminated treated wood waste" is excluded from the biomass waste list (and is considered "waste"). Wood grown on metal contaminated soil might exceed threshold values for contaminating elements and might therefore need separate treatment during combustion. If threshold values in Table 3-21 are not exceeded, the harvested wood can be considered as non-contaminated treated wood waste (and thus "biomass waste"). Additional to these legal threshold values, we should use common sense and compare metal concentrations in wood grown on contaminated land with metal concentrations in coal (see 3.2.3.2).

We cannot find a positive categorization for metal enriched biomass, *i.e.* we do not find a legislative reason why it should be considered as waste or biomass. Therefore, we recurred to elimination, *i.e.* reasons why it should not be considered waste. In doing this, some questions that should be asked are the following:

- What is the main purpose of the crop?
- What is the concentration of metals in the crop?
- What about the rest product?
- Does the elevated concentration have an effect on energy production?
- Is there an intention to dispose off the biomass?

In summary, based on current European and Flemish legislation we make the following suggestions concerning interpretation of classification of the crops used in this study. Based on the definition of waste in European and Flemish legislation, we would argue for energy crops grown on contaminated land to be considered as biomass. Several arguments exist to exclude the harvested biomass from the studied crops from being defined as waste. A first argument is that the studied crops accumulate metals in such small amounts that the main purpose of the producer (*i.e.* the farmer) is to produce energy as an alternative income generating activity within the context of sustainable land management, with remediation still being an objective, but only secondary. A second argument, as will be demonstrated later, is that energy conversion of crops exhibits no effect from the metals in the crops after harvest. Therefore, it can be stated that the purpose of the farmers is not to "dispose off" the harvest, but to economically valorize it, in the same way as would have been possible if these crops were grown on unpolluted soil. However, in Flanders, energy maize, rapeseed, and SRC of willow grown for decontamination purposes could be considered as biomass waste (depending on the interpretation of the OVAM), but this still means that they fall within the wide definition of biomass. In addition, wood waste could fall outside the scope of biomass waste and be considered as waste, depending on contamination levels in the harvested wood. Therefore, in our study, wood seems to be the only crop where there might be an actual implication of the presence of metals.

Following the example of Germany (see 3.2.2.3), chemical composition could be included as a quantification criterion to distinguish between biomass with and without elements imposing a potential burden on environment, when biomass is converted into energy. This implicitly assumes that the classification of crops will be based on the products after (as opposed to before) conversion. In the next part, we analyze current legislation on energy production to study the impact of contamination on the potential use of crops for energy conversion options.

3.2.2.1 Energy maize

For digestion, we find several articles for classification in VLAREM I that might apply, such as 2.2.3. e) for the digestion of non-dangerous waste material. However, in VLAREM II we find no restrictions and there is no specific legislation on maximum concentrations of certain elements in input[61]. However, when (biomass) waste is anaerobically digested, threshold values from the Flemish Regulation concerning Waste Prevention and –Management (VLAREA) have to be respected. VLAREA bundles the actual implementation of the Waste Decree. There is common acceptance that "dilution is not a solution for pollution", implicitly stating that values on output/rest product (digestate) have to be respected by the input. A European document with allowed input for anaerobic digestion is being developed since several years, but is however not yet finished, let alone legally binding (www.biogas-e.be). Also, on a regional level it is decided by the Flemish government that 60% of digestion input has to be related to agriculture, to be allowed to build the digester in an agricultural area (RO/2006/01). Based on the above, we conclude that the theoretical classification of energy maize as biomass waste or as biomass has no impact on energy conversion opportunities.

Digestate could be used as a secondary material in both cases. In the first case, the output would explicitly have to comply with VLAREA values, while in the second case, the digestate is not classified as waste and it could implicitly be assumed that it complies with VLAREA values.

3.2.2.2 Rapeseed

There are no quantified specifications regarding the input for production of biodiesel nor for the production of PPO. In Belgium, the European Directive 2003/30/EC[62] has been converted into the Royal Decision of March 4 2005 (K.B. Biofuels), within which "biofuel" is defined as "fluid or gaseous transport fuel based on biomass". Whether rapeseed from the Campine region is considered as biomass waste (and thus falls within the denominator of biomass) or as biomass as such, biodiesel from this rapeseed falls under this Decision (K.B. Biofuels). In

[61]It is however common practice that the Flemish Compost Organization (VLACO) performs analyses on digester input in the context of integrated chain management. Also, the Federal Public Service of Health, Food Chain Safety and Environment elaborated threshold values for pathogens in input (Sys, K., personal communication, April 2010). These are however not specifically related to metals and are not officially written down.
[62]Directive 2003/30/EC shall be repealed with effect from 1 January 2012 and be replaced by Directive 2009/28/EC.

this Decision, there are only threshold values for the output/rest product (cake) of the conversion installation. Also, biofuels are only allowed to be sold on the Belgian market when they comply with European CEN (European Committee for Standardization) standards. For Fatty Acid Methyl Esters (FAME, i.e. biodiesel) this is EN 14214 (2009). For PPO there is no CEN standard. Biodiesel that does not comply with (CEN) standards can be sold on the Belgian market only after approval from authorities and when dealers have signed a quality certificate (which guarantees the compliance with at least certain conditions). We therefore conclude that it does not matter whether rapeseed is regarded as biomass waste or biomass, the classification does not restrict energy conversion options.

3.2.2.3 Willow

Should we consider willow from the Campine region as biomass (waste), then VLAREM applies (Vanaken, 2008). VLAREM I and II consider three categories of woody biomass: (A) untreated, uncontaminated wood waste, (B) non-dangerous, treated wood waste (uncontaminated, painted, varnished and enhanced) and (C) dangerous, treated wood waste (contaminated with flame retardant, preservation, including wood waste originating from construction and demolition waste). This is a qualitative categorization.

Germany elaborated a quantitative classification for woody biomass based on amongst others metal concentrations (Table 3-20), since they acknowledge that untreated wood might also contain contamination, through background exposal or processing[63]. In Germany, certification of wood waste for use as fuel in a combustion installation is therefore based on RAL-GZ 428, with values for certain elements. For Pb, the maximum concentration for fuel use is 30 mg kg^{-1} dm (Langen, 2003). When wood is recycled and *e.g.* used in paperboard industry, maximum Pb concentration is 30 mg kg^{-1} dm and maximum Cd concentration is 2 mg kg^{-1} dm. Note that the latter concentration of 2 mg kg^{-1} dm for Cd in wood is rather low and even comparable with European threshold

[63] Ledin (1996) reported contents in mg kg^{-1} of Cd (0.8-2.9), Cu (3-5), Mn (30-80), Ni (0.5-0.7), Pb (1-2), and Zn (50-100) in stem wood from willow.

values (mg kg^{-1}) for certain vegetables, such as cucumber (1.6), asparagus (1.6), spinach (2.2), endive (3.3), and lettuce (4) (Table 1-6). Also, in Belgium, no such threshold values concerning metals in input material (wood) exist when wood is recycled in the paper industry. However, taxes on the effluent (water used during the process) are based on levels of certain contaminants (amongst which metals such as Cd) (SAPPI, personal communication, March 2009). In the paperboard industry there are no threshold values on input neither, but output has to comply with certain standards for safe use, such as EN 71-3 concerning the migration of certain elements.

Table 3-20 Maximum German concentration of certain metals in wood waste above which wood waste is considered contaminated for use as a fuel in a combustion installation (RAL-GZ 428) and in paperboard industry

Metal	*Max concentration for fuel use (mg kg^{-1} dm)*	*Max conc. for paperboard (mg kg^{-1} dm)*
Arsenic (As)	2	2
Cadmium (Cd)	-	2
Chloride (Cl)	600	600
Chrome (Cr)	30	30
Copper (Cu)	20	20
Fluoride (F)	100	100
Lead (Pb)	30	30
Mercury (Hg)	0.4	0.4

Source: Langen (2003)

On a European level, the CEN is elaborating CEN TC 335 on solid biofuels, a list of standards for pure biomass fractions (amongst other biomass waste) to be used as biofuels. Standards might include the method of determination of metals that are applicable to solid biofuels. The first part of this standardization finished in November 2010 and documents thereon are available from February 2011 (www.cen.eu). The applicability of this standardization is restricted to fractions which are covered by the LCP Directive, and fractions which are excluded (if used exclusively) from the WI Directive (www.erfo.info).

The classification of woody biomass is important for the determination of the appropriate emission directive (output). For Flanders, legislation on emissions is fixed in VLAREM I and II. VLAREM I defines the different types of installations and the obligations that follow thereof. Biomass waste falls under type 2, while other biomass falls under type 43 (Vanaken, 2008). VLAREM II handles the actual emission regulation of each type.

Within type 2, a combustion installation for biomass waste, only categories A and B are allowed (art. 5.2.3bis4). Cat. C wood waste is incinerated in combustion installations for waste (art. 5.2.3bis1) (Vanaken, 2008). Based on the maximum concentration of certain elements (Table 3-21) we decide whether wood waste is considered as uncontaminated (cat. B) or contaminated (cat. C) treated wood waste (art. 5.2.3bis4.14).

Table 3-21 Maximum concentration of certain elements below which treated wood waste is considered as uncontaminated (cat. B)

Contaminant	*a (mg kg^{-1})*†	*b (mg kg^{-1})*†
Arsenic (As)	2	4
Copper (Cu)	20	40
Lead (Pb)	90	180
Chrome (Cr)	30	60
Fluoride (F)	30	60
Chloride (Cl)	600	1,200

†Compliance measurement:1° yearly and half yearly sampling: no b-concentrations are exceeded; 2° three monthly sampling: no b-concentrations are exceeded and ¾ of the samples do not exceed a-concentrations; 3° >4 samples per year: no b-concentrations are exceeded and 80% of the samples do not exceed a-concentrations

In this study, relevant metals are Cd, Pb and Zn, only one of which can be found in Table 3-21. When Pb concentrations in wood are < 90 mg kg^{-1} dm, we consider it as cat. B wood waste, when Pb concentrations are > 90 mg kg^{-1} dm, we consider it as cat. C wood waste.

3.2.2.4 Conclusion on input

We conclude based on elimination, that metal enriched energy crops should not be considered as waste. The distinction between biomass and biomass waste is only important for wood waste, since other legislation will apply depending on the classification, having implications on types of combustion installations and emissions. For energy maize and rapeseed, there is no explicit legislation on input, but there is a close link with the output/rest product. The concentration of metals in input will determine the concentration of metals in output and the latter has to comply with threshold values. This means that we should not consider all biomass coming from the Campine region as equal, but rather analyze the output for contamination levels. The output might (in some cases very likely) comply with values for secondary use of rest products. This is described next.

3.2.3 Energy conversion of biomass grown on metal enriched soils in the Campine

3.2.3.1 Policy on emissions of metals

The Third (Den Haag, 1990) and Fourth (Esbjerg, 1995) North Sea Conference postulated a reduction in metal emissions to the air of 50-70% as compared to 1985. For 2020, a 90% reduction is targeted, for each metal separately. The Flemish Environmental Policy Plan, for 2003-2007 (MINA 3) proposed a reduction of 70% compared to 1995, which has already been accomplished for Cd in 1998 (VMM, 2008). MINA has been prolonged until 2010 (MINA 3+) to cover the period until MINA 4 starts (2011-2015). In MINA 3+ there has been no change in the goals that were set in MINA 3 regarding emissions of metals to the air. Also, the Convention on Long Range Transboundary Air Pollution Protocol demands a reduction of emissions by industry (ferro and non-ferro), energy, transport, and waste combustion. Therefore, emissions of amongst others Cd, Mercury (Hg) and Pb are registered.

In September 2005, the European Commission presented her thematic strategy on Air Pollution. On the one hand, the LCP Directive specifies conditions for combustion plants >50 MWth. This directive was integrated into VLAREM II

(Chapter 5.43). It encourages the combined generation of heat and power and sets specific emission limit values for the use of biomass as fuel. Its overall aim is to reduce emissions of acidifying pollutants, particles, and ozone precursors[64]. On the other hand, the WI Directive specifies conditions for the combustion of waste. It aims to prevent or to reduce possible negative effects on the environment caused by the incineration and co-incineration of waste. In particular, it should reduce pollution caused by emissions into the air, soil, surface- and groundwater, and thus lessen the risks which these emissions pose to human health. The WI Directive distinguishes between incineration plants (which are dedicated to the thermal treatment of waste, with or without heat recovery), and co-incineration plants (cement or lime ovens, steel plants or power plants, with energy generation or the production of material products as main purpose).

On Flemish level, VLAREM II puts restrictions on the maximum concentrations of contamination found in the biomass or waste and emissions from combustion installations that incinerate woody biomass. Art. 5.2 defines regulations for waste treatment installations. More specifically, art. 5.2.3bis4 handles (co-)combustion of (biomass) waste (cat. A and B wood), art. 5.2.3bis1 handles (co-)combustion of waste (cat. C wood), art. 5.43 covers biomass combustion not considered in art. 5.2.

On European level, concerning the control of emissions, waste gases, construction, …, category A wood falls within the LCP Directive (when > 50 MWth)**.** Category B and C fall within the scope of the WI Directive. The emissions from installations > 50 MWth that fall within the latter categories are in tune with those of large combustion plants (Vanaken, N., personal communication, March 2009). Table 3-22 gives an overview of European and Flemish legislation on emission of particular elements.

[64]The LCP Directive allows existing combustion plants to be exempted from compliance with emission limit values and from inclusion in a national emission reduction plan, provided that the operator undertakes not to operate the plant for more than 20,000 hours starting from 1 January 2008 and ending no later than 31 December 2015.

Table 3-22 Overview of emission threshold values of Cd, Tl, Hg, NO_x, SO_x, and PM (mg m^{-3}) for (co-) combustion of different categories of biomass and (biomass) waste, for different installations

Mg m^{-3}	*PM*	*NO_x*	*SO_x*	*Cd and Tl*	*Hg*	*Other metal*
WI				0.05	0.05	0.5
LCP						
Old installations (<2002)	100 (<500MWth) 50 (>500)	500-600	400-2,000			
New installations	50 (50-100) 30 (>100)	200-400	200-850			
VLAREM II waste combustion						
5.2.3bis4.9	150 (<5MWth)	400	300	-	-	-
(biomass waste,	30 (5-50)	400/200	300	-	-	-
incl. cat. A wood)	10 (>50)	200/130	50	-	-	-
5.2.3bis4.15	150 (<5)	400	300	-	-	-
(non contam.	30 (5-50)	400/200	300	0.1	0.1	1.5
treated wood=cat. B wood)	10 (>50)	200/130	50	0.05	0.05	0.5
5.2.3bis1.15 ((hazardous)waste)	10†	150-400	50	0.05	0.05	0.5
5.43 (biomass)	5.2.3bis4.9 (large) 5.2.3bis4.9 (average) 5.2.3bis4.9 (small)	5.2.3bis4.9 (large) 5.2.3bis4.9 (average) 5.2.3bis4.9 (small)	5.2.3bis4.9 (large) 5.2.3bis4.9 (average) 5.2.3bis4.9 (small)			
VLAREM II co-combustion of (hazardous) waste						
5.2.3bis1.19 +	30 (<50 MWh)			0.05	0.05	0.5
5.2.3bis1.21	10 (>50 MWh)			0.05	0.05	0.5

†daily average; Source: WI, LCP Directive, and VLAREM II

Also, the Integrated Pollution Prevention and Control Directive (96/61/EG) obliges amongst others installations which are contaminating to use Best Available Techniques (BAT) to reduce emissions. Evaluation of the use of BAT is based on Best Reference documents (BREFs), where per sector the BAT are determined and the related emission levels.

In 2005, the European Commission did a proposal for a new Directive "Ambient Air Quality and Cleaner Air for Europe". This Directive (2008/50/EC) integrates and repeals as of 11 June 2010 the Framework Directive Air (96/62/EG) and three daughter Directives (amongst which 1999/30/EG on NO_x, SO_2, Pb and

PM_{10} emissions). The fourth daughter Directive (2004/107/EG on As, Cd, Ni, Hg and PAH) might also be integrated in the future.

3.2.3.2 Effect of metals on energy production potential

Energy maize

Information, data and studies relating to the potential influence of metal concentrations in biomass on the digestion process are also scarce. Metals have a proven effect on the enzymes responsible for the break-down of biomass particles. Whether they stimulate or inhibit biogas production depends on total metal concentration, the chemical form of the metals, and process related aspects (Chen, Cheng and Creamer, 2008). Pahl *et al*. (2008) found in their experiment with co-digestion of mechanically biologically treated municipal waste containing metals and sewage sludge evidence of the accumulation of metals in the digester. According to Marchaim (1992), certain metals (not specified) can be toxic to anaerobic organisms, even at low concentrations. The metal ions affect organisms by inactivating functional groups of their enzymes and thus inhibit digestion. Wong and Cheung (1995) conducted experiments on digestion of metal contaminated sewage sludge and concluded that presence of certain metals always tends to reduce biogas yield (toxicity Cr > Ni > Cu > Zn). In contrast, studies on water hyacinth (*Eichhornia crassipes*), channel grass (*Vallisneria spiralis*) and water chestnut (*Trapa bispinnosa)* used as phytoremediating plants in industrial effluents, demonstrated that the slurry of these plants produces significantly more biogas than the slurry of control plants grown in unpolluted water (Singhal and Rai, 2003; Verma *et al.*, 2007). These experiments indicate an effect of metals on the digestion process, but they do not allow to come to an unambiguous conclusion concerning biogas production from polluted energy maize.

Therefore, a subset of each sample energy maize plant from the trial field (total plant, stem, leaves, bract, rachis, and grain) was transported to Organic Waste Systems (OWS, Ghent, Belgium) to determine its biogas production potential in a batch test. In order to investigate the difference in biogas production potential of maize grown on contaminated soil versus maize grown on uncontaminated soil, samples were compared with maize originating from an uncontaminated reference site of OWS. The small scale batch test (14 days) showed no significant difference in biogas production potential between maize grown on contaminated (215 ± 23 Nm^3 ton^{-1}) and uncontaminated (194 ± 4 Nm^3 ton^{-1}) soils. Nevertheless, further confirmation of these findings in a continuous test over a longer period of time is necessary and research is ongoing on this matter (unpublished results). Resultingly, no modifications to the biogas installation are necessary. Moreover, because all metals end up in the digestate, no modifications are needed to the CHP engine.

Rapeseed

For PPO there are no indications that the ash content after burning could pose any problems. Ash content of renewable fuels commonly lies below 1 m% and most commonly below 0.1 m%. When biomass with a high (>2%) fatty acid content is combusted, this causes corrosion. However, fatty acid content in PPO is smaller than 1%. The viscosity of PPO lies above that of diesel. Therefore, for PPO to be used in a diesel engine, the oil should be preheated prior to use, or mixed with diesel, or converted into biodiesel through esterification (Goovaerts *et al.*, 2009). In literature, we find no data on the effect of metals in biomass on the biodiesel production process, although the idea of using the remediating crops for energy purposes is gaining ground (Shi and Cai, 2009). Therefore, no modifications are necessary to the existing installation (although accurate monitoring will be necessary). Mench *et al.* (2010) point out that in order to minimise trace element emissions of vehicles running on biodiesel or PPO from crops used for contaminated soil management, crops could be selected based on low trace element contents in their seeds. Since we assume that all metals end up in the cake (unpublished results), no marginal modifications are necessary to the engine for burning PPO or biodiesel.

Willow

Possible co-combustion technologies in large electricity installations are given in Table 3-23.

Table 3-23 Co-combustion technologies in large electricity installations

Technology	*Description*
Direct co-combustion	Pretreated renewable directly injected in oven
Indirect co-combustion	Renewable is gasified, then used in installation
Parallel combustion	Combustion in separate boiler

Source: Goovaerts *et al.* (2009)

Obernberger has done extensive research on the combustion of biomass (*e.g.* Obernberger *et al.*, 1997; Obernberger, 1998). This resulted in maximum values of certain elements in biomass and renewable energy resources above which emission threshold values might be exceeded and technical problems might occur. Concentrations in Table 3-24 can be used as an indication of when potential problems might occur when biomass and other renewable energy resources are combusted.

Table 3-24 Indicative concentrations of Cd and Zn having an effect on combustion, and technical solutions

Element	*Concentration (dm%)*	*Restricting parameter*	*Technical solution*
Cd	<0.0005	Ash recycling, ash use	Partial separation of metals, ash treatment
	-	Emission of PM	Efficient PM separation, condensate treatment
Zn	<0.08	Ash recycling, ash use	Partial separation of metals, ash treatment
	-	Emission of PM	Efficient PM separation, condensate treatment

Source: Obernberger (1998); Goovaerts *et al.* (2009)

The emission of metals is closely related to the emission of particulate matter (PM). The reduction of PM emissions will thus have a direct impact on the emission of metals (Theunis *et al.*, 2003b). Best available techniques for the reduction of PM emissions are amongst other multi cyclones, fibrous filters, and electrostatic precipitators. When a multi cyclone is placed before a filter fly ash precipitator, the multi cyclone recycles the minerals from the ash fraction, while metals are separated from the fly ashes through the fly ash precipitator (Goovaerts *et al.*, 2009).

In Flanders, legislation is developed in such way that classification of harvested woody biomass as biomass or as waste has an impact on the legislation that applies on emissions during combustion. This legislation puts restrictions on maximum concentrations of contamination in the biomass and in emissions from combustion installations that incinerate woody biomass: when willow is considered as waste, it will have to be burned in an installation subject to stricter emission threshold values. SO_x emissions are lower for biomass than for coal and oil, while NO_x emissions could be higher. The latter emissions can however be reduced through primary measures and secondary reduction techniques such as flue gas cleaning (Goovaerts *et al.*, 2009). Concerning legally obliged additional flue gas cleaning, a lot depends on the legal status of the biomass, *i.e.* whether willow grown on metal contaminated land is considered as waste or not. When this willow is considered as biomass (waste), no additional flue gas cleaning will be necessary according to legislation. An example can already be found in the coal power plant of Electrabel in Ruien (Belgium) which co-combusts untreated wood. Emissions from this installation are not higher than emissions of a conventional coal installation, and the installation does not have deNO_x or deSO_x equipment. However, when willow is considered as waste, the selected installation will have to meet stricter emission threshold values. The installation we consider in this study (Langerlo) already has a deSO_x and deNO_x installation, which will make it possible to comply with stricter standards, such that additional investments will not be necessary. Flue gas desulphurization needs $CaCO_3$, but the amount of sulphur in wood is low, compared to coal. We conclude that co-combustion of wood used for phytoremediation has no marginal effect on the desulphurization process (Theunis *et al.*, 2003a).

Table 3-25 Average metal concentration in coal in mg kg^{-1}

	As	*Cd*	*Co*	*Cr*	*Cu*	*Hg*	*Mn*
Min	1.4	0.05	3.2	7.4	8.8	0.04	19
Max	18.5	0.73	8	40	16.9	0.33	200
	Ni	*Pb*	*Sb*	*Sn*	*Tl*	*V*	*Zn*
Min	7.3	4.8	0.65	25	0.6	14.8	3.2
max	40.5	32	1.6	62	0.7	43	85

Source: Theunis *et al.* (2003b)

Table 3-25 gives an overview of metal concentration ranges in coal, based on Theunis *et al.* (2003b). The averages found in Meuleman and Faaij (1997) lie within these ranges. Theunis *et al.* (2003b) conclude that in electricity installations with a $deNO_x$ electrostatic filter and a $deSO_x$ installation, total elimination efficiency of all metals (excl. Hg) >99.9%.

Theunis *et al.* (2003b) then extrapolate emission factors from coal combustion for the co-combustion of high energetic waste, and the authors conclude that when high calorific waste is co-combusted, total emissions of Cd might at worst rise with 0.97%. They attribute this raise primarily to the low Cd content in coal compared to the studied waste. Based on Theunis *et al.* (2003b) we assume in our further calculations that co-combustion of willow biomass enriched with metals does not necessarily lead to higher emissions of metals. Moreover, we can assume that most of the metals end up in the fly ashes, and these fly ashes constitute only a small fraction of total ashes.

Concerning the effect of metals on the combustion process, we made the comparison with the commonly combusted coal, which also contains metals and concluded that metals in biomass will have no marginal effect on the combustion process and on the technical efficiency of the installation (unpublished results).

3.2.3.3 Conclusion on energy conversion

Based on the above, we assume for the three crops under investigation that metals have no marginal effect on the energy conversion (technical) efficiency. Moreover, no metals end up in the energy carrier, but are concentrated in the rest products: ashes, digestate, and cake. Therefore, we studied the marginal impact of the contamination on processing the rest products for secondary use or disposal, since a different end use might have to be found for the rest products due to the contamination.

3.2.4 Marginal impact of metals on end use of rest product

3.2.4.1 Waste Policy

<u>Waste policy in Europe</u>

The European Commission proposed on 21 December 2005 a new thematic strategy on the prevention and recycling of waste (COM(2005)666). In its Communication "Taking sustainable use of resources forward: A Thematic Strategy on the prevention and recycling of waste", the European Commission makes proposals concerning the definition and classification of waste. Current EU waste policy is based on a concept known as the waste hierarchy. This means that, ideally, waste should be prevented and what cannot be prevented should be re-used, recycled or recovered (*e.g.* energy recovery) in this order as much as feasible, with disposal being used as little as possible. The legal framework for this strategic approach includes legislation on waste management such as the Waste Directive, the Hazardous Waste Directive, and the Waste Shipment Regulation. The two former directives are repealed as of 12 December 2010 and replaced by the WF Directive. We base our reasoning about the classification of digestate, cake, and ashes on this new WF Directive[65].

To decide whether products (such as digestate, ashes and cake) are to be defined as a product or a by-product, Van den bergh (2009) suggests looking at the primary aim of the process (that generates digestate, ashes and cake). This will in most or all cases be energy production, but in all cases never the production itself of digestate, ashes or cake, which leads us to conclude that these rest products are not products but fall within the category of by-products.

In a next step, we can decide whether these by-products are considered as a non-waste by-product or as waste. In the Directive, waste is defined in art. 3 (1) as *"any substance or object which the holder discards, intends to discard or is required to discard"*[66]. To clarify the concept of discard, the European Union uses

[65]Complementary, there is detailed legislation concerning waste treatment and disposal operations in the Landfill (99/31/EC) and Incineration Directives, and there is legislation to regulate the management of specific waste streams (COM(2005)666).
[66]This is the same definition as used in the Waste Decree.

the European Waste Catalogue established by Decision 2000/532/EC of the European Commission and largely taken over by VLAREA in Flanders, or makes references to other concepts such as "disposal" and "recovery" (Van den bergh, 2009).

Concerning by-products, art. 5 of the WF Directive says the following "*a substance or object, resulting from a production process, the primary aim of which is not the production of that item, may be regarded as not being waste but as being a by-product only if the following conditions are met: (a) further use of the substance or object is certain".* This can be interpreted as the use being useful and having a financial advantage for the user (Van den bergh, 2009) and "*(b) the substance or object can be used directly without any further processing other than normal industrial practice"* Van den bergh (2009) states that a by-product should be regarded as waste if it cannot be used in a normal industrial practice without recovery. "Recovery" is in art. 3 (5) of the WF Directive defined as an operation that prepares waste to fulfill a useful function to replace other materials. *"(c) the substance or object is produced as an integral part of a production process".* It should not leave the plant for further processing before being usefully used*; "and (d) further use is lawful, i.e. the substance or object fulfils all relevant product, environmental and health protection requirements for the specific use and will not lead to overall adverse environmental or human health impacts."*

If digestate, ashes and cake are classified as waste (and not as a by-product), we could determine if they apply for an end-of-waste status (art. 6, WF). Concerning the end-of-waste status of products and materials the Directive says the following: *"Certain specified waste shall cease to be waste when it has undergone a recovery, including recycling, operation and complies with specific criteria to be developed in accordance with the following conditions: (a) the substance or object is commonly used for specific purposes; (b) a market or demand exists for such a substance or object; (c) the substance or object fulfils the technical requirements for the specific purposes and meets the existing legislation and standards applicable to products; and (d) the use of the* *substance or object will not lead to overall adverse environmental or human health impacts."*

The WF Directive sets the basic concepts and definitions related to waste management and introduces waste management principles[67]. The WF Directive lays down measures to protect the environment and human health by preventing or reducing the adverse impacts of the generation and management of waste, by reducing overall impacts of resource use, and by improving the efficiency of such use. Therefore, any incineration or co-incineration should recover energy at a high level of efficiency. The current WF Directive defines recovery and disposal of waste taking into account the fact that energy produced by a municipal incinerator substitutes the use of resources in other power plants such that the definition will better reflect the environmental benefits of incineration. Recovery is defined as *"any operation the principal result of which is waste serving a useful purpose by replacing other materials which would otherwise have been used to fulfill a particular function, or waste being prepared to fulfill that function, in the plant or in the wider economy".* Disposal is defined as *"any operation which is not recovery even where the operation has as a secondary consequence the reclamation of substances or energy (WF, art. 3)".* Previously, the overwhelming majority of municipal incinerators were considered as disposal facilities.

Also, when digestate, ashes and cake are classified as waste, we could determine whether they are classified as hazardous or non-hazardous waste. In the former case, more stringent regulation will apply concerning treatment, storage and transport (Van den bergh, 2009). For a product to be defined as hazardous waste (art. 41, WF), it should be mentioned in the waste list of the Waste Catalogue and have hazardous properties (as described in Annex III of the WF).

[67]Not included in this directive are (amongst others) [ ] land (in situ) including unexcavated contaminated soil and buildings permanently connected with land [ ].

Concerning costs, art. 14 of the WF Directive states that costs of waste management shall be borne by the original waste producer or by the current or previous waste holders. Member States may decide that costs of waste management are to be borne partly or completely by the producer of the product from which the waste came, and that distributors of such product may share these costs.

Directive 99/31/EC on the landfill of waste aims to prevent or reduce as far as possible negative effects on the environment from the landfilling of waste, by introducing stringent technical requirements for waste and landfills. It defines the different categories of waste (municipal waste, hazardous waste, non-hazardous waste, and inert waste) and applies to all landfills, defined as waste disposal sites for the deposit of waste onto or into land.

<u>Waste policy in Flanders</u>

Waste management in Flanders as known today took its final shape with the Waste Decree, through which the OVAM was established. The fundamentals of this agency reside in the Ladder of Lansink. The Ladder of Lansink assumes the following order when handling waste: prevention, reuse, recycling, composting, combustion with energy recovery, combustion without energy recovery, and landfilling. The complete waste policy is described in MINA 3 under the topic "contamination through waste material". At the core of this policy are 4 goals: (1) the production of waste and the pressure on the environment is not allowed to increase, (2) natural resources and energy should be replaced by waste material as much as environmentally can be justified, (3) landfill and combustion of waste material should be avoided as much as environmentally justifiable, and (4) the effectiveness of waste policy will be elevated. The ultimate goal is to limit landfilling to non-reusable and non-combustible materials such as material containing asbestos. This policy is confirmed in MINA 3+ (2008-2010).

VLAREA (based on the WF Directive) lists up all waste material and categorizes them. The appendices to VLAREA sum up values for the secondary use of waste material (Table 3-26). Waste material can be used

(1) as or in fertilizer or soil improving material;

(2) as or in construction material; and
(3) as soil.

Table 3-26 Conditions for use of rest products as a secondary material based on origin of these rest products

Rest product	*Origin*	*Conditions*
Use as or in fertilizer or soil improving material		
Digestate	Certified installation for (anaerobic) digestion of waste (combined with manure)	Table 3-28 + VLACO
Meal from cake	Pressing plant oils	Table 3-28
Use as or in construction material		
Fly + bottom ash	Ferro- and non-ferro industry	Table 3-33 + certificate
	Combustion of waste†	Table 3-33 + certificate
	Combustion	Table 3-33

†According to Theunis *et al.* (2003b) it depends on the interpretation of the VLAREA appendix whether bottom- and fly ashes could be reused as a secondary material. They argue that wood waste combustion installations do not fall within VLAREA, but that when they are considered as installations for household waste, VLAREA does apply and if the ashes comply with the values, they could be reused. We argue that (i) wood waste combustion installations always fall within the scope of VLAREA and (ii) recalling art. 5.2.3.4.1. § 1 of VLAREM II, wood of cat. B ís considered as being combusted in household waste combustion installations, and VLAREA applies.

3.2.4.2 Secondary use of digestate

Digestate originating from organic biological waste (animal and plant) is listed in VLAREA under category 19 06 06. It can be applied as soil improving material and is not considered as a waste by-product (that has to be disposed off) when it complies with VLAREA values regarding total concentrations of certain elements and total soil application of these elements (Table 3-28), and when it receives a certificate after investigation of the Flemish Compost Organization (VLACO). The Action Plan on Organic Biological Waste also stimulates digestion for the conversion of waste into soil improving material. General agricultural and environmental regulations are to be respected.

Digestion does not lead to a reduction in metals (Table 3-27). Metals can evaporate when bound to other matter, but to a low degree. As the organic fraction of the biomass is broken down during digestion (and metals are not), the relative fraction of metals will increase (Kool *et al.*, 2005). The concentration of metals in the digestate depends on the ratio between organic matter and water content of the biomass in the input. For maize and cow manure, the mass reduction is 50-75% and 3% respectively (Velghe, F., personal communication, September 2007). On the experimental farm De Marke in Hengelo (The Netherlands) the concentration of metals in energy maize from normal, undiluted soil was measured (row 3). In row (1) the values of metals in energy maize of a sample in the Campine are indicated. We elaborated a digestion model for energy maize in Chapter 4.2.

Table 3-27 Average metal concentration in maize, digestate (mg kg^{-1} dm)

	Maize	*Cadmium*	*Zinc*	*Lead*
(1)	Enriched maize (30 dm%)[†]	0.65 - 1.35	186 - 301	2.40 - 4.20
(2)	Enriched digestate (11 dm%)[‡]	2.25 - 4.64	643.85 - 1,041.92	8.55 - 14.61
(3)	Regular maize (30 dm%)	0.11	37	0.4
(4)	Regular digestate (11 dm%)	0.38	128.08	1.38

[†]Table 3-1; [‡]1 ha maize (30 dm%): 18 ton dm + 42 ton water, 1 ha digestate (11 dm%): 42 + 11% x = x, which leads to x= 47.2 and 11% x= 5.2. The metal concentration in the digestate triples (5.2/18), which is consistent with findings in Van Slycken *et al.* (xxxx); [‡]Kool (2006)

There does not exist such thing as "the" digestate. In general, digestate is a good alternative to chemical fertilizer and in addition is more stable than undigested manure, with a better humus performance (Timmerman *et al.*, 2005). The digestate (11 dm%) could (i) be applied on the land as a replacement for chemical fertilizer. This means however in our study that metals are re-applied on the land. This could be justified with the new concept of phytoattenuation (Meers *et al.*, 2010) and within the context of energy maize as a contributor to sustainable land management in the Campine region (Thewys *et al.*, 2010a; 2010b). Spreading the metals on the Campine land would not dilute them. Taking into account values from Table 3-28 (VLAREA), the maximum amount of digestate that could be reapplied on the land is 2.80 ton per hectare per year (Zn being the limiting metal: 1,800/643.85).

Although the reapplication of metal enriched digestate might seem unproductive and controversial, the re-application of contaminated sludge has several examples. As Mirck *et al*. (2005) put it: "*Where biomass production and pollutant management overlap, the science of phytoremediation has its practical application."* Near Enköping (Sweden) farmers cultivate *Salix Viminalis* clones that are co-combusted (20%) in a district heating power plant. This willow has been used as a vegetation filter in several applications. In one of the applications, residual ash from the power plant is mixed with an equal amount of treated sludge from a local wastewater treatment plant. This mixture contains metals, and in a representative sample the following concentrations can be found: Cd, 0.75; Cu, 194.5; Cr, 26.1; Hg, 0,33; Ni, 12.9; Pb, 15; Zn, 324 mg l^{-1}. This mixture is then delivered, free of charge, to the local farmers to spread on their willow land as a fertilizer. More examples can be found in Mirck *et al.* (2005).

When digestate is (ii) traded to spread on other land than the originating land[68], maximum trace element concentrations are indicated in the K.B. Fertilizer. These values are stricter than for own use (Table 3-28, col. 3). Based on comparison of concentrations of Cd, Zn, and Pb in digestate in Table 3-27, and maximum allowed concentrations of Cd, Zn, and Pb in digestate in Table 3-28 we conclude that values for Cd and Zn are exceeded, leading to the conclusion that export to other countries or use on other land is not an option.

A third option is (iii) processing the digestate. The digestate is separated, dried and subsequently combusted, or disposed. Manure is already combusted with a combustion value of 14-19 MJ ton^{-1} dm in England, The Netherlands, and Scotland (Lemmens *et al.*, 2007). In the current case study, the residual digestate is further processed by separation into a liquid fraction (2 dm%) and a solid sludge fraction (30 dm%). The liquid fraction is disposed off, given that it complies with VLAREM II threshold values (art. 5.28.3.5§3 and appendix 5.3.2.24bis). The solid fraction is dried (85 dm%) using the heat recuperated

[68]The Royal Decision contains a list of products that can be traded as fertilizer or soil improving material. Digestate is not in this list, but could be traded by sending a request to the Federal Public Service of Health, Food Chain Safety and Environment.

from the CHP unit powered by the biogas generated by the anaerobic digestion process. Preliminary test results of OWS show that around 3% of the metals will end up in the liquid fraction (Peene, 2009). Final results are forthcoming (Van Slycken *et al.*, xxxx).

Table 3-28 VLAREA values regarding (1) total concentrations of certain elements, (2) total soil application of these elements, and (3) total concentrations for export of digestate and rapeseed meal

Metals	*Total concentration own land (mg kg^{-1}dm)*	*Total soil application own land (g ha^{-1} y^{-1})†*	*Total concentrations export (mg kg^{-1} dm)*
	(1)	(2)	(3)
Arsenic (As)	150	300	-
Cadmium (Cd)	6	12	1.5
Chrome (Cr)	250	500	-
Copper (Cu)	375	750	-
Mercury (Hg)	5	10	-
Lead (Pb)	300	600	120
Nickel (Ni)	50	100	-
Zinc (Zn)	900	1,800	300

†Maximum soil application depends on soil characteristics such as pH and organic material

3.2.4.3 Secondary use of rapeseed cake

In VLAREA, cake from rapeseed pressing is not mentioned as a waste material. However, meal is mentioned and could be used as soil improving material when it complies with VLAREA standards regarding total concentrations of certain elements and total soil application of these elements (Table 3-28). Rapeseed could also be combusted, replacing light fuel oil, if reuse would not be possible, respecting the Ladder of Lansink. When rapeseed is pressed to PPO, the dm% of the cake stays the same, implying that the metal concentration has not changed compared to the raw material (Table 3-2).

Cake (after cold pressing) can (i) be used on a farm to replace barley and fodder (Suenens, 2007). Rapeseed cake has also been (ii) digested in a batch test by OWS (dry digestion). These tests showed a large biogas potential but the cake is nitrate rich (C/N ratio of 9.1 while 20 is preferred) (Peene, 2009). The yield of cake lies between 460-600 m³ ton^{-1} fm (Torrijos *et al.*, 2008; Peene, 2009). Other options for secondary end use are (iii) combustion of cake, and (iv) landfill.

3.2.4.4 Secondary use of ashes

Rest products are inevitable and are produced in power plants as a result of requirements to meet air emission standards set in EC Directives, like the LCP Directive, and as a result of plant design to meet product standards, or both. Each of the rest products has specific physical and chemical properties that make them suitable for utilization in established markets (Eurelectric, 2006; Nielsen, 2008).

Regardless of the combustion technology, rest products after combustion (of both coal and biomass) are bottom ashes, fly ashes and rest products of flue gas cleaning (Obernberger *et al.*, 1997) (Table 3-29). Bottom ashes are uncombusted minerals which end up on the bottom of the boiler, often mixed with mineral impurities contained in the biomass fuel as well as with sintered ash particles (when the combustion temperature exceeds the ash temperature, the ash melts and results into slag). Fly ash is uncombusted material that leaves the boiler with the flue gases. The boiler and cyclone fly ashes are the fine, mainly inorganic, ash particles carried with the flue gas and precipitated in the boiler section and in (multi-)cyclones placed behind the combustion unit. The filter fly ash is the finer fly ash fraction precipitated in electrostatic filters, fibrous filters or as condensation sludge in flue gas condensation units (normally placed behind the multi-cyclone). In small biomass combustion plants this ash fraction is emitted with the flue gas. The flue dust is the finest fly ash fraction, not precipitated due to its small particle size and therefore emitted with the flue gas. Compared to oil and natural gas, solid fuels like biomass and coal contain much more ash forming material. Desulphurization of flue gases results in lime or plaster, avoiding dioxin emissions uses active coal which has to be captured with a filter (Theunis *et al.*, 2003a; Goovaerts *et al.*, 2009). Table 3-30 shows average concentrations of relevant metals in ashes from virgin biomass.

Table 3-29 Percentage of the different ash fractions to the total ash amount (m%)

Biomass fuel/ash fraction	*Bark*	*Wood chips*	*Straw and cereals*
Bottom ash	75.0-85.0	70.0-90.0	80.0-90.0
Boiler and cyclone fly ash	15.0-25.0	10.0-30.0	3.0- 6.0
Filter fly ash	2.0- 5.0	4.0- 8.0	5.0-10.0
Flue dust	0.1- 2.0	0.2- 3.0	0.2- 1.0

Source: Obernberger *et al.* (1997)

Table 3-30 Concentration ranges of combustion relevant elements in ashes from different virgin biomass (mg kg^{-1} dm)

	Wood chips (spruce)	*Bark (spruce)*	*Straw (wheat, rye, barley)*	*Cereals (wheat, tritic.)*
Zn	260-500	300-940	60-90	120-200
Cd	3.0-6.6	1.5-6.3	0.1-0.9	0.1-0.8

Source: Obernberger (1998)

Meuleman and Faaij (1997) present an overview of the percentage of metals in each of the rest products in a conventional coal combustion installation. Cd and Pb are mostly found in fly ashes (Table 3-31). The desulphurization installation eliminates the last emissions from the flue gases. This can then be compared with the percentage of metals in each of the rest products in a biomass combustion installation (Table 3-32).

Table 3-31 Average distribution of As, Cd, Hg, and Pb among different components in a conventional coal combustion installation (%)

Different components	*As*	*Cd*	*Hg*	*Pb*
Total	100	100	100	100
Bottom ash	1.6	1.8	-	3.1
Fly ash	88	95	17	102
Flue gas deSO$_x$ installation	0.4	3.5	28	0.9
Flue gas	-	-	56	-

Source: Meuleman and Faaij (1997) and Theunis *et al.* (2003b)

Table 3-32 Average distribution of Cd, Pb, and Zn among different ash fractions in a biomass combustion plant (%)

Ash fraction	*Cd*	*Pb*	*Zn*
Bottom ash	3.4	9.8	11.1
Cyclone fly ash	54	35.4	43.8
Filter fly ash	42.7	54.8	45.1

Source: Narodoslawsky and Obernberger (1996)

Based on Table 3-29, Table 3-31, and Table 3-32 we can confirm the findings from Obernberger *et al.* (1997) that it rewards to separately handle filter fly ash (low m%, but high concentration of metals). They can then subsequently either be disposed to landfill or industrially utilized (metal recovery).

In VLAREA, the rest products (fly- and bottom ash) coming from electricity installations and other combustion installations is categorized under 10 01. When waste is combusted, the rest product is considered under 19 01. Rest products from zinc metallurgical processes are categorized under 10 05.

The conditions for use of waste material as secondary material (construction) in Flanders are described in VLAREA (Table 3-33). Exceptionally however, even if conditions in column (1) or (2) are not met, waste might be used as a secondary construction material (granulates). To use granulates as a secondary material, an additional user certificate is necessary.

Table 3-33 VLAREA values regarding total concentrations, leaching factor, and maximum immission of metals in ashes, for the ashes to be used as a secondary material

Metals	*Total concentrations (mg kg^{-1} dm)*[†]	*Leaching factor for reuse as non-formed construction material (mg kg^{-1} dm)*[‡]	*Maximum immission (mg m^{-2} over 100 years)*[§]
	(1)	(2)	(3)
Arsenic (As)	250	0.8	285
Cadmium (Cd)	10	0.03	12
Chrome (Cr)	1,250	0.5	555
Copper (Cu)	375	0.5	255
Mercury (Hg)	5	0.02	8.2
Lead (Pb)	1,250	1.3	609
Nickel (Ni)	250	0.75	136
Zinc (Zn)	1,250	2.8	924

[†]Target values. When this number lies below background values (VLAREBO) leaching does not have to be determined; [‡]Obligatory values, waste that complies with values in column (2) can be used in formed construction material; [§]When used as formed construction material, column (2) must result in column (3)

Applications include, amongst others, use in cement (as both raw oven feed material and as a direct cement replacement), in concrete, for the production of lightweight aggregates and blocks, as aggregates in building and road industries, in mining and other operations as a construction or fill material, as mineral fillers, and as raw material in the gypsum industry for the production of plasterboard (Eurelectric, 2006; Nielsen, 2008). Only products that followed the procedures of CEN and have an EC mark can enter the market. The European standards (EN-) for cement and concrete do not specify limitations on the metals concentration and leaching, but EN-197 mentions that there are countries in which these limits exist, and these will remain in force, also when European standards will be applied (Cenni *et al*., 2001).

On a European level there is no regulation concerning the reuse of bottom ashes resulting from waste combustion[69]. For the production of blended cement, fly

[69]More specifically, regarding the reuse of granulates in landfill (end layer, drainage layer, …) legislation should make some things more clear. Now, the reuse in landfill is seen as

ash has to meet European standard EN-197-1. Fly ash can be added to concrete to enhance its technical performance for a number of reasons (EN-206-15, and when replacing cement EN-450-16 and EN-450-27). Fly ash is used as a siliceous source in the manufacture of aerated concrete blocks (EN-771) and it has also been used as raw material in the manufacture of lightweight aggregates (EN-13055). Fly ash could also be used for the production of bricks and mortar according to national legislation and requirements (Eurelectric, 2006). Bottom ash is the preferred material by all manufacturers due to the lightweight nature and stability of the aggregate. Bottom ash used as a coarse and fine aggregate in the manufacture of concrete blocks has to meet the requirements of EN-13055 and national regulations. It can also be used as a raw material for cement clinker production (site specific requirements), as filler for cement (EN-197-13), and for brick production (national regulations). Fly ash, bottom ash and boiler slag are used in a number of applications as aggregates in building and road construction (European and national standards) (Eurelectric, 2006).

Ashes from combusting metal enriched biomass from the Campine region cannot be used as granulates, since they surpass VLAREA threshold values (Table 3-33). This conclusion is based on concentration levels of Cd in willow (Table 3-3), and the percentage of metals ending up in the ashes (Table 3-31).

3.2.4.5 Conclusion on legislation rest products

Table 3-34 gives an overview of theoretical possibilities for utilization of rest products from biomass harvested on a metal enriched field. Hazardous waste disposal (landfill) is an option when no other option is allowed according to legislation.

---

useful reuse if bottom ash complies with VLAREA norms. If however granulates are used as support or leveling in a landfill, it is seen as landfill and not as a useful end use (Nielsen *et al.*, 2008).

Table 3-34 Overview of potential end use of rest products for different scenarios, based on input category (waste or biomass), and on legislation for end product use

Input	*Conversion process*	*Rest product*	*Input category*	*Rest product category*	*Secondary use*
Willow	Co-comb. electricity	Fly+Bottom ash	Waste/Biomass	(10 01)	Construction + landfill†
	Co-comb. heat	Fly+Bottom ash	Waste/Biomass	(10 05)	Construction + landfill†
	Co-comb. CHP	Fly+Bottom ash	Waste	(19 01)	Construction + landfill†
			Biomass	(10 01)	Construction + landfill†
Maize	Anaerobic digestion	Digestate	Waste	(19 06 05/06)	Fertilizer (11 dm%)‡ Combustion (85 dm%) Landfill (85 dm%)
			Biomass	-	Fertilizer (11 dm%)‡ Export (85 dm%)‡[70] Combustion (85 dm%) Landfill (85 dm%)
Rapeseed	PPO	Cake	Waste/Biomass	-	Fodder (89 dm%)§ Digestion (89 dm%) Combustion (89 dm%) Landfill (89 dm%)
	Biodiesel	Cake and glycerin	Waste/Biomass	-	Fodder (89 dm%)§ Digestion (89 dm%) Combustion (89 dm%) Landfill (89 dm%)

†Table 3-33; ‡on own land, and Table 3-28 ; §Directive 2002/32/EC, and Fodder Decision: maximum level of Cd for fodder in Belgium (and Europe) expressed as mg kg^{-1} dm is 1.14

[70]In practice we will not include export as an option for uncontaminated digestate as the digestate we are studying does not include manure, and there is thus no obligation for elimination.

Chapter 3.3 *Economic analysis of rest products*

Parts of this chapter have been submitted in:
Witters, N., Van Slycken, S., Weyens, N., Meers, E., Tack, F., Vangronsveld, J. (xxxx) Does phytoremediation redeem to the expectations of being a sustainable remediation technology: a case study II: economic assessment and policy analysis of CO_2 abatement. Biomass Bioenerg.

Van Slycken, S., Croes, S., Guisson, R., Witters, N., Meers, E., Adriaensen, K., Michels, E., Ruttens, A., Vangronsveld, J., De Jonghe, W., Thewys, T., Tack, F.M.G. (xxxx) Exploring the phytoextraction potential of Cadmium and Zinc on metal contaminated agricultural soils using Brassica napus L. Environ. Pollut.

Abstract

The presence of metals in the residue after conversion of biomass is a scholarly example of an externality as this fact is not reflected in the price of the original biomass. We will provide a transparent and comprehensive environmental and economic evaluation of a range of strategies for rest products. The aim of this section is to economically valorize the effect of metals in the residue, to be integrated in the total economic value of phytoremediation. The economic impact of metals on the end use of rest products is based on current legislation. More specifically, the value of this externality is calculated based on a comparison of options for the contaminated residue with the best available (economically most efficient) reference option for uncontaminated residue, based on the market price and MAC CO_2.

When biomass is uncontaminated, and applying existing legislation, digestate from digestion of energy maize should be used as a fertilizer, with the remaining digestate being combusted. Rapeseed cake should be used as fodder, and ashes from willow should be valorized as granulates. For contaminated biomass, strictly applying current legislation, applying part of the digestate on the land and combusting the rest, results in the highest total economic value. For cake, the use as fodder results in no marginal impact from metals. Willow ashes have to be disposed off because metal concentration is too high.

3.3.1 Introduction

When studying the secondary use of rest products after conversion of biomass crops used for plant-based technologies, we suggest not following blind folded the obliged order of the waste management hierarchy as defined in the European Union (and the Ladder of Lansink in Flanders). Alternatively, we suggest to use a Cost-Benefit Analysis (preferably accompanied by a Life Cycle Analysis) and choose the option that generates the largest social benefit. This was also acknowledged in preparing the Waste Framework (WF) Directive: *"The waste hierarchy generally lays down a priority order of what constitutes the best overall environmental option in waste legislation and policy, while departing from such hierarchy may be necessary for specific waste streams when justified for reasons of, inter alia, technical feasibility, economic viability and environmental protection."*

Metals are an example of stock pollutants, pollutants that accumulate over time. Environment has little or no absorptive capacity for them. It is not possible to destroy metals, but we should aim to find a solution so that they do not pose a threat to current and future generations. We offer a systematic approach to find the economically most efficient and cost-effective solution for biomass produced on contaminated land. We started from existing regulation on secondary use of rest products of energy production to value the internalization of the presence of metals. By strictly applying the rules, the presence of metals in the biomass is internalized. This leads to economic efficiency if these rules and threshold values are correctly based on *e.g.* health impact studies.

"Economic instruments can play a crucial role in the achievement of waste prevention and management objectives. Waste often has value as a resource, and the further application of economic instruments may maximise environmental benefits. The use of such instruments at the appropriate level should therefore be encouraged while stressing that individual Member States can decide on their use" (WF Directive).

3.3.2 *Data and methods*

3.3.2.1 Metals in biomass as an externality of plant-based technologies

We studied the effect of metals in biomass on the emission control of the conversion installations. For rapeseed pressing and anaerobic digestion we concluded that there was no effect at all, as compared to general biomass conversion. In our calculations, the separation and drying technology are available at the digestion installation. For the combustion of willow, we compared with coal, cokes, and other waste and decided that the marginal effect on metal emissions of replacing coal, cokes, and other waste with contaminated willow would be negligible. The considered combustion installations have a $deNO_x$ electrostatic filter, a $deSO_x$ installation, and a desulphurization installation[71]. This means that the potential marginal effect of metals in the biomass focuses on the impact on secondary use of the rest product after energy conversion.

We valorize the marginal effect of metals on the potential end use of the rest product through application of existing legislation regarding energy production and waste disposal. Energy production using crops grown on metal contaminated land results in metal enriched rest products. This might cause a raise in costs or reduction in benefits to avoid that metals become a danger for human health. This valorization is a correct approach and will lead to an economically efficient outcome under the assumption that current regulation is based on health- and environmental studies, resulting in correct economic values. Then by applying regulation on rest products, we assure that negative effects of metals in rest products on health and environment are completely and correctly internalized.

An example of a policy that fully internalizes negative health effects can be found in the immission levels (particulate matter) for Cd in Flanders (5 ng Cd m^{-3} to be realized in 2012), regulated by VLAREM II. This and other threshold values on air quality originate from a European Directive relating to Cd in

[71]An extended flue gas cleaning would require an additional investment of € 293 ton^{-1} input (Nielsen *et al.*, 2008).

ambient air (Rosier, A., personal communication, October 2009). This Directive is based on a health impact study of the European Commission (EC, 2001) and on World Health Organisation (WHO) Directives. If a member country fails to live up to these values, the Commission takes action concerning condemnation (Claeys, N., personal communication, December 2009). By enforcing these threshold levels, the negative health impact of Cd is correctly internalized.

In internalizing externalities we follow several steps as defined by Bickel and Friedrich (2005). First, we define the different options for the metal enriched rest products. This is based on integrating technical data regarding concentrations in the rest products, and legislation on allowed concentrations for the end use of rest products (Table 3-34). Second, the valuation of these options is based on (i) the market price of the secondary uses, and (ii) MAC CO_2. Third, the economic values of metal enriched residues are then compared with the economic values of the best available uncontaminated reference options. The marginal effect of metals in rest products is then determined as the difference between both values.

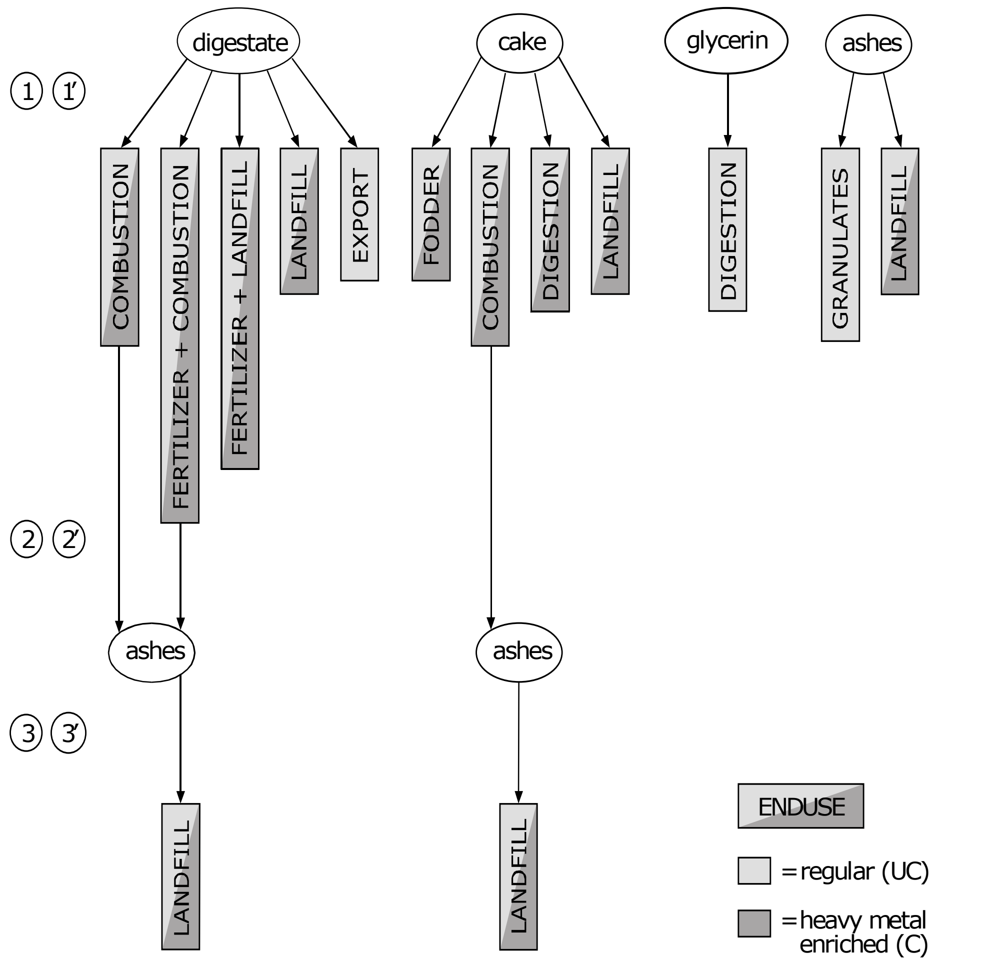
The presence of metals is also valued through the marginal abatement cost (MAC) of CO_2[72]. The logic behind this is that when metals have an impact on the potential end use of rest products (such as combustion of digestate instead of using it as fertilizer), this might very likely have an impact on potential CO_2 abatement and hence should be part of the economic analysis when internalizing the externality of metals.

3.3.2.2 General approach

Figure 3-2 gives an overview of the different steps to internalize the externality of metals in the rest product. The different options for end use of the rest products in an uncontaminated (UC) and contaminated (C) scenario are based on Table 3-34. Some of the options (when combustion is included) result in an

[72] In Chapter 3.5 we discuss the concept of MAC CO_2 in detail.

additional rest product, ashes, for which an end use has to be found. Glycerin results when the rapeseed is not pressed to PPO, but to biodiesel (Table 3-4).



1 and 1′ Energy input for end use of the rest product (incl. transport), respectively valued through MAC CO_2 and market price

2 and 2′ Energy output or avoided energy use due to the end use of the rest product, respectively valued through MAC CO_2 and market price

3 and 3′ Transport and landfill cost of ashes respectively valued through MAC CO_2 and market price (no energy cost for landfill)

Figure 3-2 Marginal effect of metals: metal enriched rest products compared to regular biomass rest products

The approach is the same for each of the remediating crops (energy maize, willow, and rapeseed). In Chapter 3.4 we study the CO_2 abatement potential of biomass for energy, compared to fossil energy. The energy budget is thus based on the energy potential of the uncontaminated rest product. Previously, we decided that on the trial field and with our studied crops, metals have no direct effect on energy efficiency as such, but that there might be an indirect effect of metals in the rest product on potential CO_2 abatement, due to restricted options for the rest products (see 3.2.3 and 3.2.4). In this chapter, we calculate the necessary marginal changes due to the presence of metals in each of the crops.

$E_{post\ in\ i\ C}$ = energy input for end use (i) of contaminated rest product

$E_{post\ out\ i\ C}$ = energy output by end use (i) of contaminated rest product

$E_{post\ out\ i\ C} - E_{post\ in\ i\ C} = E_{post\ i\ C}$ = net energy after end use (i) of contaminated rest product

$E_{post\ i\ C} \cdot MAC\ CO_2 = V_{EN\ post\ i\ C}$ = economic value of net energy after end use (i) of contaminated rest product based on CO_2 abatement

$M_{post\ i\ C}$ = amount of alternative product (substitute) that is replaced by the contaminated rest product, *e.g.* fertilizer, roughage, cokes, … by its end use (i)

$M_{post\ i\ C} \cdot p_{post\ i\ C} = V_{M\ post\ i\ C}$ = economic value of contaminated rest product after end use (i) based on market price

$V_{post\ i\ C} = V_{EN\ post\ i\ C} + V_{M\ post\ i\ C}$ = economic value of contaminated rest product after end use (i) based on the sum of CO_2 abatement and market price

$V'_{post\ C}$ = best end use option of contaminated rest product based on economic value based on CO_2 abatement and market price substitute

= max ($V_{post\ i\ C}$) (if $V > 0$)

= min ($V_{post\ i\ C}$) (if $V < 0$)

The same definitions apply for uncontaminated biomass (UC). The end use that is economically most viable for general rest products after energy conversion is the one that generates $V'_{post\ UC}$. As a result, the external effect of metals on the rest product based on CO_2 abatement and market price substitute is $EV_{post} = V'_{post\ C} - V'_{post\ UC}$

If $EV_{post} > 0$: external benefit of metals on rest product based on CO_2 abatement and market price (EB_M)
If $EV_{post} < 0$: external cost of metals on rest product based on CO_2 abatement and market price (EC_M)

3.3.2.3 Assumptions in the economic analysis

Table 3-35 gives an overview of parameters and values used in calculations of secondary use of digestate, cake, and ashes.

Table 3-35 Overview of parameters and values used in calculations of secondary use of digestate (DIG), cake and ashes

Parameter	*Value*		*Source*
Prices			
MAC CO_2	20	€ ton^{-1}	Bickel and Friedrich (2005)
Fertilizer	150	€ ha^{-1}	Aegten,pers comm (April 2010)
Cokes	90	€ ton^{-1}	www.e-coal.com
Water disposal	0	€ ton^{-1}	
Natural gas	23.39	€ MWh^{-1}	www.creg.be
Electricity	27.78	€ GJ^{-1}	www.creg.be
Boiler Efficiency			
Cokes	80	%	CHP ref Decision (2006)
Natural gas	90	%	CHP ref Decision (2006)
Energy out- and input			
CH_4	35.9	MJ m^{-3}	Dalgaard *et al*. (2001)
DIG separation-electricity	27-36	MJ m^{-3} input	Lemmens *et al*. (2007)
DIG drying-heat	2.37	GJ ton^{-1} solid	Lemmens *et al.* (2007)
Cokes	29	GJ ton^{-1}	www.emis.vito.be
CO_2 emission (100%)			
Natural gas	202	g kWh^{-1}	IPCC (2007)
Diesel	267	g kWh^{-1}	IPCC (2007)
Electricity average	115	kg GJ^{-1}	MIRA (2008b)
Cokes	107	kg GJ^{-1}	IPCC (2007)

MAC CO_2 = marginal abatement cost CO_2, DIG = digestate

3.3.3 Results

3.3.3.1 Economic valuation of digestate

Table 3-36 summarizes the different stages that digestate passes through after digestion. For use as a fertilizer on own land a dry weight percentage of 11 dm% is sufficient, while for combustion or landfilling, the digestate will first need to be separated (until 30 dm%) and dried until 85 dm%. Other options have not yet been studied in Flanders.

Table 3-36 Liquid, solid, and total portion of energy maize and different stages of digestate (ton ha^{-1})

Stage of energy maize	*Liquid (ton ha^{-1})*	*Solid (ton ha^{-1})*	*Total (ton ha^{-1})*
Energy maize (30 dm%)	42	18	60
Digested maize (11 dm%)	42	5.2	47.2
Digestate separated (30 dm%)	12.1	5.2	17.3
Digestate after drying (85 dm%)	0.9	5.2	6.1

We calculate the value of digestate per hectare (€ ha^{-1}) and per ton (€ ton^{-1}), valorized by the market price ("market"), and by MAC CO_2 ("CO_2"), for combustion, landfill, and fertilizer use respectively (Table 3-37, Table 3-38, Table 3-39, and Table 3-40). Table 3-42 and Table 3-43 then give an overview for respectively valorization through MAC CO_2 and through the market price.

Combusting digestate results in 12.73 GJ ha^{-1} (Table 3-37). This translates into a net energy output of 0.27 GJ ton^{-1} digestate (11 dm%). This is consistent with general results in Flanders, where the combustion of digestate is an energetic zero operation (Sys, K., personal communication, March 2010). Per hectare 1.7 ton CO_2 emission is avoided, and per ton digestate (11 dm%) 40 kg.

The private cost (based on price of cokes and electricity) to dry and separate digestate (11 dm%) to digestate of 85 dm% is € 3.52 ton^{-1} digestate (11 dm%). The social cost (based on CO_2 emissions) to dry and separate digestate (11 dm%) to digestate of 85 dm% is € 2.41 ton^{-1} digestate (11 dm%). However, the net benefit is positive. Separating and drying and subsequenty combusting the digestate translates into a net private economic benefit of 49.4 € per hectare (€ 1.05 per ton digestate of 11 dm%) due to the production of energy, and a net social economic benefit 34.7 € per hectare (€ 0.74 per ton digestate 11 dm%) due to avoided CO_2 emissions.

Table 3-37 Net benefit per hectare and per ton when digestate (DIG) (11 dm %) is combusted to replace for cokes (Nyrstar) (negative values between brackets)

	GJ		*Ton CO_2*		*CO_2 (€)*		*Market (€)*	
	ha^{-1}	ton^{-1} DIG	$ha^{-1¶}$	ton^{-1} DIG	*$ha^{-1\#}$*	ton^{-1} DIG	*$ha^{-1††}$*	ton^{-1} DIG
Gross heat	55.51†	1.18	7.42	0.16	148.49	3.15	215.34	4.56
Heat use	(41.08)‡	(0.87)	(5.49)	(0.12)	(109.89)	(2.33)	(6.59)	(0.14)
Elec. use	(1.70)§	(0.04)	(0.20)	(0)	(3.90)	(0.08)	(159.36)	(3.38)
Net heat	14.43	0.31	1.93	0.04	38.60	0.82	55.98	(1.19)
Net elec.	(1.70)	(0.04)	(0.20)	(0)	(3.90)	(0.08)	(6.59)	(0.14)
Net DIG	12.73	0.27	1.73	0.04	34.70	0.74	49.38	1.05

†6.1 ton digestate: (i) 5.2 ton is combusted at 0.35 m³ CH_4 kg^{-1} COD (chemical oxygen demand), the amount of oxygen for combustion; for proteins this is 1.1-1.4; for fats and oils this is 2-2.7; for digestate we use 0.85 (η(th) =78% *1.1= 0.85), (ii) during combustion 0.9 ton water is evaporated at an energy cost which is completely recuperated by condensation, as is often the case (Sys, K., personal communication, March 2010); ‡drying 17.3 ton solid digestate (30 dm%) at 2.37 GJ ton^{-1}; §separating 47.2 ton digestate (11 dm%) at 36 MJ ton^{-1}; ¶heat from cokes, average electricity; #MAC CO_2 = € 20 ton^{-1}; ††heat from cokes (€ 90 ton^{-1}, 29 GJ ton^{-1}, 80% boiler efficiency)

In Table 3-38, we present landfill costs per ton as of 2007 (OVAM, 2008c). The difference between the high and low cost is due to taxes, which is € 40 ton^{-1} for cat. 1 waste (hazardous), and € 75 ton^{-1} for cat. 2 waste non-hazardous industrial). Combustible waste is subject to a high tax rate, to enforce the order of the Ladder of Lansink. Waste from soil remediation operations is not subject to environmental taxes when landfilled. Depending on the interpretation of the Waste Decree, digestate (and cake of rapeseed as well) could be regarded as waste from a soil remediation operation. Regardless, we calculated with both values (with and without tax).

Prior to landfill, the digestate (11 dm%) is separated and dried. The private cost to dry and separate digestate (11 dm%) to digestate of 85 dm% is € 3.52 ton^{-1}

digestate (11 dm%), while the social cost to dry and separate digestate (11 dm%) to digestate of 85 dm% is € 2.41 ton^{-1} digestate (11 dm%). These costs are lower than what it would have cost to landfill 47.2 ton instead of 6.1 ton digestate. However, landfilling still remains a costly option, despite the prior drying of the digestate, with costs depending on the landfill category, and whether or not taxes are included (Table 3-39).

Table 3-38 Landfill costs (cat. 1 and cat. 2) in 2007 (€ ton^{-1})

	Cat. 1 (€ ton^{-1})	*Cat. 2 (€ ton^{-1})*
Without tax	53	39
With tax	93	114

Source: OVAM (2008c)

Table 3-39 Net benefit per hectare and per ton when digestate (11 dm%) is landfilled (after separation and drying), without (w/o) and with tax

	CO_2 (€)				*Market (€)*			
	ha^{-1}		ton^{-1}		ha^{-1}		ton^{-1}	
	Cat. 1	Cat. 2	Cat. 1	Cat. 2	Cat. 1	Cat. 2	Cat. 1	Cat. 2
W/o tax	(438.0)	(352.4)	(9.3)	(7.5)	(490.2)	(404.5)	(10.4)	(8.6)
With tax	(682.7)	(811.2)	(14.5)	(17.2)	(734.9)	(863.4)	(15.6)	(18.3)

When digestate is applied on the land, metals are returned to the land. However, the allowed amount of digestate is small (Manure Decree) and therefore also the resulting amount of metals reapplied on the field.

In Flanders, Council Directive 91/676/EEC on the protection of water against nitrates coming from agricultural sources is converted into the Manure Decree (MD[73]). Art. 3 of the MD points out that digestate coming from digestion with manure is considered as animal fertilizer, whereas digestate coming from digestion without manure is considered as other fertilizer. This has implications for the application of the digestate. When energy maize is digested without manure (*i.e.* dry digestion), it can be applied in addition to the maximum amount of allowed manure, while making sure not to exceed the maximum

[73]Often referred to as Manure Action Plan (MAP) III.

allowed sum of animal manure and other fertilizer. From January 2009, the maximum amount of N and P_2O_5 coming from manure on maize land is respectively 170 and 85 kg P_2O_5 ha^{-1} $year^{-1}$. The same amount of digestate coming from dry digestion of energy maize can be applied additionally, but the sum of N and P_2O_5 cannot exceed respectively 275 and 85 kg ha^{-1} $year^{-1}$ (Manure Decree).

One hectare generates 60 ton fm of energy maize, resulting in 47.2 ton (11 dm%) digestate. The destination of 47.2 ton digestate depends on the contamination (UC versus C) (Table 3-41). In the uncontaminated scenario, MD applies, while in the contaminated scenario both MD and VLAREA apply. 1 ton maize contains 4 kg N and 0.6 ton P_2O_5. 1 ha of land with maize can at maximum take 170 kg N and 85 kg P_2O_5. In the uncontaminated scenario, when digestate is reapplied, N is the limiting element. 170 kg N equals 42.5 ton maize equals 33.43 ton digestate (11 dm%), following the same logic as in Table 3-36. The rest of the P_2O_5 (=85-20 kg) and 105 kg N could be applied with manure. However, in the contaminated scenario, because of metal concentrations in the digestate (Table 3-27) and VLAREA values in Table 3-28 (Zn being the limiting metal), the maximum allowed amount of digestate per ha is 2.80 ton, and lies far below the maximum amount allowed by the MD (33.43 ton) in the uncontaminated scenario (UC).

Table 3-40 Net benefit per hectare and per ton when digestate (11 dm%) is used as a replacement for fertilizer (N) on own land†

	Kg N ha^{-1}	*Ton dig ha^{-1}*	*CO_2 (€)*		*Market (€)*	
			ha^{-1}	ton^{-1}	ha^{-1}	ton^{-1}
UC	170‡	33.3	13.42	0.40	150	4.49
C	14§	2.8	1.12	0.40	12.56	4.49

UC = uncontaminated, C = contaminated; †based on West and Marland (2002): 55.53 GJ ton^{-1} N, 3,947 kg CO_2 ton^{-1} N; ‡Manure Decree; §1 ton maize generates 4 kg N

The production of fertilizer is energy consuming and requires 57 GJ ton^{-1} N, and due to the use of cokes and electricity, almost 4 ton CO_2 is emitted per ton N (West and Marland, 2002). When 33.43 ton digestate (11 dm%) is reapplied on the land, the production of 170 kg N is avoided. Moreover, no chemical fertilizer has to be bought, resulting in a benefit of € 150 ha^{-1} (Table 3-40). When 2.8 ton digestate (11 dm%) is reapplied on the land, production of 14 kg N can be avoided. The resulting benefit per hectare is lower, while the cost per ton applied digestate (11 dm%) remains the same as in the uncontaminated scenario.

The rest of the digestate has to be subsequently separated (until 30 dm%), dried (until 85 dm%) and combusted or landfilled (Table 3-41). The costs per ton digestate (11 dm%) and per hectare based on the market price and on the CO_2 emission can be found in Table 3-37 for combustion, in Table 3-39 for landfill, and in Table 3-40 for fertilizer use.

Table 3-41 Amount of digestate (liquid and solid fraction) in ton per ha destined for combustion (1), and for fertilizer use combined with combustion for uncontaminated (2) and contaminated (3) digestate

	UC/C‡		*UC*§		*C*¶	
	(1)		(2)		(3)	
Fertilizer	0		33.43		2.8	
Combustion	47.2		13.77		44.4	
	Liquid	*Solid*	*Liquid*	*Solid*	*Liquid*	*Solid*
Digestate (11 dm%)	42	5.2	12.26	1.51	39.52	4.88
Digestate (30 dm%)	12.11	5.2	3.53	1.51	11.40	4.88
Digestate (85%)†	0.92	5.2	0.27	1.51	0.86	4.88

†Only the solid fraction will actually result in net energy output. The liquid fraction needs energy to evaporate, but given complete energy recuperation through condensation, the liquid fraction results in a zero energy change (Sys, K., personal communication, March 2010); ‡Table 3-36; §based on Manure Decree; ¶based on Manure Decree and VLAREA

Economic valuations in Table 3-42 are similar to those in Table 3-43, but based on market prices instead of MAC CO_2. MD applies when digestate is uncontaminated, while VLAREA applies additionally to contaminated digestate.

Table 3-42 OVERVIEW Value of digestate (11 dm%) when applied on own land, exported, combusted or landfilled, based on market price (negative values between brackets)

Options	*€ ton^{-1} digestate (11 dm%)*		*€ ha^{-1}*	
Export	-		-	
COM	1.05		49.38	
LF 1	(15.57)–(10.39)		(734.89)–(490.18)	
LF 2	(18.29)–(8.57)		(863.37)–(404.54)	
	MD	VLAREA	MD	VLAREA
FER + COM	3.48	1.25	164.40	59.00
FER + LF 1	(1.36)–0.15	(14.38)–(9.5)	(64.34)–7.03	(678.82)–(448.61)
FER + LF 2	(2.16)–0.68	(16.94)–(7.80)	(101.81)–32.01	(799.68)–(368.04)

COM= combustion, FER= fertilizer, LF1= landfill 1, LF2= landfill 2

Table 3-43 OVERVIEW Value of digestate (11 dm%) when applied on own land, exported, combusted or landfilled, based on MAC CO_2 (negative values between brackets)

Options	*€ ton^{-1} digestate (11 dm%)*		*€ ha^{-1}*	
Export	-		-	
COM	0.74		34.70	
LF 1	(14.46) – (9.28)		(682.73) – (438.02)	
LF 2	(17.19) – (7.47)		(811.20) – (352.38)	
	MD	VLAREA	MD	VLAREA
FER + COM	0.50	0.72	23.54	33.77
FER + LF 1	(3.93)–(2.42)	(13.58)–(8.71)	(185.72)–(114.35)	(641.17)-(410.96)
FER + LF 2	(4.73)–(1.89)	(16.14)–(7.00)	(223.19)–(89.37)	(762.03)–(330.38)

COM= combustion, FER= fertilizer, LF1= landfill 1, LF2= landfill 2

An additional effect of metals in digestate is the disposal of the liquid fraction when digestate is separated prior to drying. The amount of metals that end up in the liquid fraction is very small (see 3.2.4.2), so we assume no extra costs for extra treatment of the liquid fraction. However, because of the metals, options for digestate change, having an impact on the amount of liquid fraction that needs to be disposed off at a cost of € 5 ton^{-1} (Sys, K., personal communication, April 2010). There are no effects on CO_2 emission abatement. Table 3-44 is based on Table 3-41. In "combustion" and "landfill" options, 29.89 ton of liquid fraction have to be disposed off. There is no difference in liquid disposal costs between uncontaminated and contaminated digestate because the presence of metals puts no restrictions on combustion or landfill of digestate, and thus the same amount of digestate needs separation prior to combustion. In the "fertilizer + combustion" and "fertilizer + landfill" options 8.72 ton and 28.12 ton liquid fraction have to be disposed off in the uncontaminated and contaminated scenario respectively.

Table 3-44 Liquid fraction disposal costs after separation (€ per ha) in (i) "combustion" and "landfill" options, and (ii) in "fertilizer + combustion" and "fertilizer + landfill" options, for uncontaminated (UC) and contaminated (C) digestate

	(i)		(ii)	
	UC	*C*	*UC*	*C*
Amount of water (ton)	29.89	29.89	8.72	28.12
Cost liquid disposal (€ ha^{-1})	149.44	149.44	43.61	140.60
Cost liq disposal (€ ton^{-1} 11 dm%)	3.17	3.17	3.17	3.17

Table 3-45 is similar to Table 3-42, but including disposal of the liquid fraction. This makes the "combustion" option lag even farther behind the "fertilizer + combustion" option. For uncontaminated digestate, the Manure Decree applies, and "fertilizer + combustion" is the option where the rest product generates a benefit. Contaminated digestate (VLAREA), no longer generates a positive revenue, and "fertilizer + combustion" is the best option.

Table 3-45 OVERVIEW Value of digestate when applied on own land, exported, combusted or landfilled, based on market prices, including disposal of liquid fraction (negative values between brackets)

Options	*€ ton^{-1} digestate (11 dm%)*		*€ ha^{-1}*	
Export	-		-	
COM	(2.12)		(100.06)	
LF1	(18.74) – (13.56)		(884.33) – (639.62)	
LF2	(21.46) – (11.74)		(1012.81) – (553.98)	
	MD	VLAREA	MD	VLAREA
FER+COM	0.31	(1.92)	120.79	(81.60)
FER+LF1	(4.53)–(3.02)	(17.55)–(12.67)	(107.95)–(36.58)	(819.42)–(589.21)
FER+LF2	(5.33)–(2.49)	(20.11)–(10.97)	(145.42)–(11.60)	(940.28)–(508.64)

COM= combustion, FER= fertilizer, LF1= landfill 1, LF2= landfill 2

3.3.3.2 Economic valuation of rapeseed cake

Table 3-46 summarizes the different stages of rapeseed after pressing. Rapeseed and cake have equal dm%. No pre-processing of the cake will be necessary prior to fodder use (Table 3-47), digestion (Table 3-48), combustion (Table 3-49) or landfill. Table 3-52 provides an overview of all options, valued through the market price as well as through MAC CO_2.

Table 3-46 Rest product after pressing of 1 ha rapeseed

State of rapeseed	*Liquid (ton ha^{-1})*	*Solid (ton ha^{-1})*	*Total (ton ha^{-1})*
Rapeseed (90 dm%)	0.3	3	3.33
Cake PPO (89 dm%)	0.23	1.93	2.16[†]
Cake biodiesel (89 dm%)	0.21	1.72	1.93[†]

[†]3.33 ton fm seeds minus 35 m% conversion efficiency to PPO, or minus 42 m% conversion efficiency to biodiesel

We included the use of cake to replace fodder because the average concentration of metals in rapeseed cake does not exceed European threshold values for metals in fodder.

Table 3-47 Net benefit per hectare and per ton (89 dm%) valued by the market price and MAC CO_2 for cake (a) after biodiesel and (b) after PPO, used as fodder for (1) dairy cattle, and (2) beef cattle

	Original fodder				*Value Cake*					
	Cost		*Cost with cake*				(a)	(b)	(a)	(b)
	(1)	(2)	(1)	(2)	(1)	(2)	(1)		(2)	
	€ $year^{-1}$				*€ ton^{-1}*		*€ ha^{-1}*			
Market	9,216	1,854	4,654	745	156	152	302	338	294	329
	GJ $year^{-1}$				*€ ton^{-1}*		*€ ha^{-1}*			
CO_2	46.55	9.98	19.95	3.33	1.35	1.35	2.61	2.92	2.61	2.92

In Table 3-47, on a dairy cattle farm 36.5 ton roughage (€ 212 ton^{-1}) + 14.6 ton barley (€ 100 ton^{-1}) could be replaced by 29.2 ton cake + 21.9 ton roughage, on a beef cattle farm 7.3 ton roughage + 3.65 ton barley could be replaced by 7.3 ton cake + 3.65 ton roughage (for extended calculations, see Suenens, 2007). This results in lower fodder costs for both farms (since the cost of the cake is already included in the cost of pressing PPO). For the calculation of CO_2 abatement, we assumed a diesel use of 9.11 GJ ha^{-1} for barley and roughage, and a yield of 10 ton ha^{-1} of these crops. These revenues can then be recalculated to a per ton revenue and per hectare revenue, based on Table 3-46.

Table 3-48 Net benefit per hectare and per ton for cake (a) after biodiesel and (b) after PPO, anaerobically digested (incl. CHP)

	Gross	*Electr.*	*Heat*	*CO_2*	*CO_2 (€)*		*Market (€)*	
	GJ ha^{-1}	*GJ ha^{-1}*	*GJ ha^{-1}*	*Ton ha^{-1}*	*ha^{-1}*	*ton^{-1}*	*ha^{-1}¶*	*ton^{-1}*
Cake (a)	16.98†	6.28‡	5.35‡	1.05§	21.08	10.91	213.09	110.33
Cake (b)	19.02	7.04	5.99	1.18	23.63	10.91	238.81	110.33

†460 m^3 gas for cake (Peene, 2009), 53.25% CH_4, 35.9 MJ m^{-3} CH_4; ‡η(el) = 40%, η(th) = 45%, β(el) = 7.5%, β(th) = 30% (η = conversion efficiency, β = use during process); §CHP replaces separate production of average electricity and heat from natural gas; ¶electricity: € 27.78 GJ^{-1}, natural gas: € 7.22 GJ^{-1} (incl. 90% boiler efficiency)

Table 3-49 Net benefit per hectare and per ton for cake (a) after biodiesel and (b) after PPO, combusted to replace for cokes (Nyrstar)

	Heat		*CO_2 (ton)*		*CO_2 (€)*		*Market (€)*	
	GJ ha^{-1}	*GJ ton^{-1}*	*ha^{-1}‡*	*ton^{-1} cake*	*ha^{-1}§*	*ton^{-1}*	*ha^{-1}¶*	*ton^{-1}*
Cake (a)	38.43†	19.90	5.14	2.66	102.79	53.22	119.25	61.74
Cake (b)	43.06	19.90	5.76	2.66	115.20	53.22	133.65	61.74

†1.93 (2.16) ton cake: 1.72 (1.93) ton is combusted with 0.35 m³ CH_4 kg^{-1} COD, COD= chemical oxygen demand, the amount of oxygen needed to combust. For cake we calculate with 1.78 (η(th) =78%*2.3= 1.78), 35.88 MJ m^{-3} CH_4 (Sys, K., personal communication, March 2010), the value of 19.90 GJ ton^{-1} dm lies below the value used by Borjesson (1996); ‡cokes (80% boiler efficiency) = 134 kg CO_2 GJ^{-1}, average electricity; §MAC CO_2= € 20 ton^{-1}; ¶cokes: € 0.09 kg^{-1}, 29 GJ ton^{-1}, 80% boiler efficiency

Glycerin is the second rest product when rapeseed is converted to biodiesel. 1 ton PPO + 0.1 ton methanol result in 1 ton biodiesel + 0.1 ton glycerin. 3.33 ton seeds result in 1.4 ton PPO. This PPO is converted to biodiesel (1.4 ton) and glycerin (0.14 ton). Glycerin is an excellent input material for digestion (750 m³ gas per ton glycerin). There is no impact of metals on the quality or quantity of glycerin. Additionally digesting glycerin after converting rapeseed into biodiesel would result in an additional revenue of € 2.41 based on MAC CO_2 and € 22.28 based on market price (Table 3-50). Landfilling of cake is a costly option, with costs depending on the landfill category, and whether or not taxes are included (Table 3-51).

Table 3-50 Net benefit per hectare and per ton for glycerin after biodiesel, anaerobically digested (incl. CHP)

	Gross	*Electr.*	*Heat*	*CO_2*	*CO_2 (€)*		*Market (€)*	
	GJ ha^{-1}	*GJ ha^{-1}*	*GJ ha^{-1}*	*Ton ha^{-1}*	*ha^{-1}*	*ton^{-1}*	*ha^{-1}*	*ton^{-1}*
Glycerin	1.94	0.72	0.61	0.120	2.41	1.25	22.28	11.54

Table 3-51 Net cost per hectare and per ton when cake (89 dm%) after biodiesel (a) and after PPO (b) is landfilled, without (w/o) and with tax

	(a)				*(b)*			
	ha^{-1}		*ton^{-1}*		*ha^{-1}*		*ton^{-1}*	
	Cat 1	*Cat 2*	*C 1*	*C 2*	*C 1*	*C 2*	*C 1*	*C 2*
W/o tax	(102.36)	(75.32)	(53)	(39)	(114.72)	(84.42)	(53)	(39)
With tax	(179.62)	(220.18)	(93)	(114)	(201.30)	(246.75)	(93)	(114)

Table 3-52 OVERVIEW Value of rapeseed cake after (a) biodiesel and (b) PPO, when used as fodder, digested, combusted or landfilled based on market prices and MAC CO_2, including glycerin (negative values between brackets)

Options	*Market* € ha^{-1}		*CO_2* € ha^{-1}	
	(a)	(b)	(a)	(b)
Fodder (dairy)	324.06	338.20	5.02	2.92
Fodder (beef)	315.85	329.00	5.02	2.92
Biogas	235.37	238.81	23.49	23.63
Combustion	141.55	133.65	105.21	115.20
Landfill cat. 1	(157.34)-(80.08)	(201.3)-(114.72)	-†	
Landfill cat. 2	(197.90)-(53.04)	(246.75)-(84.42)	-	

†No energy costs

For rapeseed the choice between conversion into (a) biodiesel or (b) PPO might depend on the biogas production of glycerin (Table 3-52). Rapeseed cake can best be used as fodder regardless whether the cake is contaminated or uncontaminated. There is thus no marginal effect of metals on the value of the rest product of rapeseed coming form metal enriched soils.

3.3.3.3 Economic valuation of willow ashes

The (co-) combustion of wood waste in Flanders in a certified installation and with recuperation of energy is not subject to an environmental tax (OVAM, 2008c). Dry ash content (m%) of untreated willow and willow chips ranges between 1.2 and 1.6. For coal this lies around 14m% (www.ecn.nl). The combustion of willow of 4.8 ton (dm) per ha leads to 57.6-76.8 kg ashes per year (we use 76.8 kg).

Bottom ashes could be exported, currently from three combustion installations in Belgium. Since 2008, ashes no longer need to be converted into VLAREA conform end products within the country of destination to be allowed for export to that country. Bottom ashes that are not destined for reuse or recycling can be landfilled immediately (cat. 1 or 2) (Nielsen *et al.*, 2008; Nielsen, 2008). Since January 2007, the chapter on environmental tariffs and taxes (chapter IX) of the

Waste Decree has been simplified (OVAM, 2008c). The tariff and tax for ash disposal can be found in Table 3-53. Table 3-56 gives an overview of legally acceptable options for ashes.

As of end 2007, in Flanders most bottom ashes are disposed at a cat. 2 landfill for non-hazardous industrial waste (73,000 out of 89,820 ton), whereas most fly ashes are disposed at a cat. 1 landfill for hazardous waste (49,435 out of 55,018 ton) (OVAM, 2008c). There are no quantitative guidelines to decide whether waste is considered hazardous. To be defined as hazardous waste (art. 41, WF), waste should be mentioned in the waste list (Waste Catalogue) and have hazardous properties (as described in Annex III of the WF).

Table 3-53 Tariffs and environmental taxes for landfill of bottom- and fly ash in Flanders and bottom ashes in European countries (€ ton^{-1})

Flanders (€ ton^{-1})			*European Countries¶ (€ ton^{-1})*	
cat1†	cat2‡	Tax§		Landfill tariff
42 (2006)	16 (2006)	0	Denmark	60-70
39 (2005)	20 (2005)		France	60-70
44 (2004)	25 (2004)		Germany	50-75
			The Netherlands	50
			Wallonia	40-45

†OVAM (2008c) for cat. 1, this is an average of all wastes because there are no separate data on bottom ashes; ‡Specific tariff for combustion ashes, lower tariff because they can serve as a layer in landfill; §Nielsen *et al.* (2008), in general, taxes have already been paid by combustion installations, so to avoid double taxation there is never an environmental tax on ash disposal; ¶Nielsen *et al*. (2008)

Bottom ashes could be upgraded to a secondary material, using different technologies, with resulting end products that can be economically valorized (granulates, sand, sludge, and a rest fraction). First, in Flanders, there are two bottom ash treatment installations: Indaver (Beveren) and Valomac (Grimbergen). Ferro (10%) and non-ferro (1%) metals that leave Indaver are reused as a secondary material. Granulates (6-50 mm: 24% and 2-6 mm: 11%) could be used as building material for foundation of roads and other constructions. Sand (0.67-2 mm: 33%) is used in construction- and stability applications of landfills. 10% is sludge (<0.67 mm). The rest fraction (8%) is landfilled. Ashes at Valomac which are not VLAREA conform are exported to Wallonia for reuse (if conform regional threshold values). Bottom ashes not destined for reuse or recycling are landfilled (cat. 1 or 2) (Nielsen *et al.*, 2008; Nielsen, 2008). Table 3-54 shows treatment costs of bottom ashes prior to secondary use. Table 3-55 shows prices for treated bottom ashes.

Table 3-54 Bottom ash: treatment costs in different European countries (€ ton-1 input)

Country	*Cost*	*Country*	*Cost*
Austria	63	Germany	25-30
Denmark	34	Italy	75
France	13-18	Luxembourg	16

Source: Nielsen *et al.* (2008)

Table 3-55 Treated bottom ash: prices in different European countries (€ ton^{-1}) (negative prices between brackets)

France	*The Netherlands*	*Sweden*
0.8 - 3.8 (50%)	(8) (special cat.)	6 (granulates)
0 (25%)	(5) – 0 (cat. 2)	2.2 (sand)
0 + transport (25%)	6 (cat. 1)	
(6.1)-(22.1) (worst case)		

Source: Nielsen *et al.* (2008)

The price of the ferro fraction is based on the world market price, and is influenced by impurities (landfill costs are deducted), oxidation level, transport, and the local market. Market prices can be lower with € 30 ton^{-1} and higher with € 5 ton^{-1}. The non-ferro fraction is sold to a secondary treatment installation. The price of this fraction is based on a standard composition (Al: 60%, and Cu, Zn, brass and inox: 25%). Aluminum content (and thus quality) determines the price (Nielsen *et al.*, 2008). Prices of metals as of February 2010 are (in € ton^{-1}) 2,323 for Cd, 2,195 for Pb, and 2,222 for Zn.

Another option is the removal of metals from the ashes in the combustion installation. Thermodynamic equilibrium calculations showed that Cd could be removed from sewage sludge ash in the form of gaseous hydroxide above 700°C. Pb is completely removed in the form of gaseous oxide above 850°C

(Fraissler *et al.*, 2009). The high volatility of some metals has already been shown by Mattenberger *et al.* (2008). Up to 90% of Cd could be removed through flue gas cleaning. The basic principle of metal separation is precipitation of the cyclone fly ash at very high temperatures, followed by effective fly ash precipitation operating at as low a temperature as possible. The decision whether to reuse ashes will depend on the cost of this separation, the value of the ashes in a secondary use and the cost of land filling. After separation, ashes could be recycled to the land. The calculated cost of Cd removal from ashes varies between € 8 and 40 ton^{-1} ash[74]. The lower cost refers to large heating plants, while the higher cost refers to small heating plants. Before application to soil, ashes must be stabilized (*e.g.* through granulation), for technical and environmental reasons. The cost of stabilizing, transporting and spreading the ash has been estimated to be, on average, about € 28 ton^{-1} ash (Borjesson, 1999b; Berndes *et al.*, 2004).

In our calculations, uncontaminated ashes (76.8 kg) are used as granulates at a cost of € 10 ton^{-1}, leading to a cost of € 0.77 ha^{-1}. When landfilled, costs range between € 1.23 and € 3.22 ha^{-1}, depending on the landfill category.

Table 3-56 OVERVIEW Value of bottom ashes when used as granulates or landfilled, valued by the market price (negative values between brackets)

Options	*Value ashes (€ ton^{-1})*	*Value ashes (€ ha^{-1})*
Granulates†	(97) – (10)	(7.45) – (0.77)
Landfill cat. 1‡	(42)	(3.22)
Landfill cat. 2‡	(16)	(1.23)

†Based on Table 3-54 and Table 3-55: (75)+(22) and (16)+6, ‡Table 3-53

3.3.3.4 Destination of ashes after combustion of rest products

Three of the previous options for rest products need additional treatment: (i) combusting digestate, (ii) using part of the digestate as fertilizer and combusting the rest, and (iii) combusting rapeseed cake. All three secondary

[74]Average euro dollar exchange rate in 2004 of € 0.8 $\$^{-1}$

uses result in an additional rest product: ash. We did not yet include the costs of depositing these final ashes. In Table 3-57 we give an overview of costs for depositing final ashes after combustion of rest products.

Table 3-57 Cost (€ ha^{-1}) of depositing final ashes (4 m%) after combustion (COM) for contaminated (C) and uncontaminated (UC) biomass, valued by the market price of landfill

		Input COM (ton)	*Ashes (ton)*¶	*Cat. 1 (€ ha^{-1})*#	*Cat. 2 (€ ha^{-1})*#
(1)	Digestate - COM (C and UC)	6.12†	0.24	10.28	3.92
(2)	Digestate – F + COM (C)	5.76‡	0.23	9.67	3.68
(3)	Digestate – F + COM (UC)	1.78‡	0.07	3.00	1.14
(4)	Cake (a) - COM (C and UC)	1.93§	0.08	3.24	1.39
(5)	Cake (b) - COM (C and UC)	2.16§	0.09	3.64	1.24

COM = combustion, F = fertilizer, UC = uncontaminated, C = contaminated; †Table 3-36; ‡Table 3-41; §Table 3-46; ¶based on Borjesson (1996); #Table 3-53: € 42 ton^{-1} for cat. 1 and € 16 ton^{-1} for cat. 2

For digestate, we notice a small effect of metals when part of the digestate is spread on the land and the rest is combusted (compare (2) and (3)). This is because the metal content determines the maximum amount that can be applied on the land, and thus determines the rest amount that will be combusted, Under the assumption that ash disposal costs are equal for ashes coming from the combustion of contaminated and uncontaminated biomass, there is no effect of metals when all digestate is combusted (1), and when all cake is combusted (4 and 5).

3.3.3.5 Transport of rest products, glycerin and ashes

The market price for transport (T in € ton^{-1}) is determined by (Eq. 7). Transport costs are also valued through their impact on CO_2 emissions. We use a diesel energy use of 1.3 MJ ton^{-1} km^{-1}, a CO_2 emission coefficient of 74.1 kg CO_2 GJ^{-1} and a price of € 20 ton^{-1} CO_2. This results in transport costs (€ ha^{-1}) of rest products and ashes (Table 3-58).

Table 3-58 Cost (€ ha^{-1}) of transport of rest products to secondary use and ashes to landfill valued by market price and MAC CO_2[‡]

	Contaminated				Uncontaminated			
	Rest product		*Ashes*		*Rest product*		*Ashes*	
Digestate								
COM	12.59 (12.24+0.34)		0.50 (0.49+0.01)		12.59 (12.24+0.34)		0.50 (0.49+0.01)	
LF cat. 1	12.59				12.59			
LF cat. 2	12.59				12.59			
F+COM	11.95 (11.51+0.44)		0.47 (0.46+0.01)		4.96 (3.57+1.39)		0.15 (0.14+0.01)	
F + LF 1	11.95				4.96			
F + LF 2	11.95				4.96			
Cake[†]	(a)	(b)	(a)	(b)	(a)	(b)	(a)	(b)
FO (dairy)	39.07 (38.63+0.45)	0			39.07	0		
FO (beef)	39.07	0			39.07	0		
DIG	3.97 (3.86+0.11)	4.45 (4.33+0.13)			3.97	4.45		
COM	3.97	4.45	0.16 (0.15+0.01)	0.18 (0.17+0.01)	3.97	4.45	0.16	0.18
LF cat. 1	3.97	4.45			3.97	4.45		
Lf cat. 2	3.97	4.45			3.97	4.45		
Glycerin	(a)				(a)			
	0.29 (0.28+0.01)				0.29			
Ashes								
	0.16 (0.15+0.01)				0.16 (0.15 + 0.01)			

COM=combustion, LF 1=landfill cat. 1, LF 2=landfill cat. 2, F=fertilizer, FO=fodder, DIG=digestion; [†](a) cake after biodiesel, (b) cake after PPO; [‡]between brackets: first number is based on market price, second on MAC CO_2

3.3.4 Discussion and conclusion

More than 20 years ago, Nelson *et al.* (1987) stated that there will always be a need to dispose off residues in the environment because some toxic constituents cannot be destroyed, or are not commercially interesting. Therefore, the volume of hazardous waste should be minimized and disposal technologies should be optimized to prevent negative environmental consequences.

In this part, we studied the marginal effect of metals on the value of the harvested biomass, based on existing threshold values. Metals are an example of stock pollutants, pollutants that accumulate over time. Environment has little or no absorptive capacity for them, as opposed to flow or fund pollutants. These stock pollutants can create a burden for future generations by passing on a damage cost which persists well after benefits received from incurring that damage cost have been forgotten (Tietenberg, 2003). It is not possible to destroy metals, but we should aim to find a solution so that they do not pose a threat to current and future generations. This chapter aims to assist, by offering a systematic approach to find the economically most efficient and cost-effective solution for biomass used for plant-based management of metal enriched soils. We started from existing regulation on secondary use of rest products of energy production to value the internalization of the presence of metals. By strictly applying the rules, the presence of metals in the biomass is internalized. This only leads to economic efficiency when we assume that these rules and threshold values are based on *e.g.* health impact studies.

When biomass is uncontaminated (Table 3-59), and following existing legislation, digestate should be used as a fertilizer, with the remaining digestate being combusted. Adding Table 3-43, Table 3-45, and Table 3-58, this option results in the highest sum of private and social benefit per hectare. Rapeseed should be used as fodder (Table 3-52, and Table 3-58), and ashes from willow should be used as granulates (Table 3-56, and Table 3-58).

Table 3-59 Economically most viable end use options for the rest product (ton ha^{-1} year^{-1})† in the uncontaminated and contaminated scenario

Rest product	*Uncontaminated*	*Contaminated*
Energy maize		
Digestate (47.2)	Fertilizer (33.43)+Combustion (13.77)	Fertilizer (2.80)+Combustion (44.40)
Ashes	Landfill (0.07)	Landfill (0.23)
Rapeseed		
Cake PPO (2.16)	Fodder (2.16)	Fodder (2.16)
Cake Biodiesel (1.93)	Fodder (1.93)	Fodder (1.93)
Glycerin	Digestion (0.14)	Digestion (0.14)
Willow		
Ashes (0.0768)	Granulates (0.0768)	Landfill (0.0768)

†Chapter 3.3

For contaminated biomass (Table 3-59) strictly applying current legislation, applying part of the digestate on the land and combusting the rest results in the highest sum of private and social value (Table 3-43, Table 3-45, and Table 3-58). Rapeseed cake should still be used as fodder (Table 3-52, and Table 3-58). Willow ashes have to be disposed off because metal concentration is too high. The impact in economic terms is however low (Table 3-56, and Table 3-58).

By comparing the economically most viable uncontaminated option with the economically most viable contaminated option, applying current legislation and valuation through market prices and CO_2 abatement, we are able to calculate the marginal impact of the elevated presence of metals (for accumulation levels specific in our case) on the economic value of the rest product (Table 3-60).

Table 3-60 Marginal impact of metals on the value of the rest product (€ ha^{-1} y^{-1}), comparing the economically most viable uncontaminated option with the economically most viable contaminated option, applying current legislation and valuation through market prices and CO_2 abatement, including transport of rest product, ashes, and glycerin, and disposal of final ashes†

			Substitute	*D_a*	*T_b*	*$T_{a,g}$*	*€ ha^{-1} y^{-1}*
Uncontaminated							
Energy maize	F + COM	Market	120.79	(3)	(3.57)	(0.14)	114.08
		CO_2	23.54		(1.39)	(0.01)	22.14
Rapeseed (a)‡	FO	Market	324.06		(38.63)	(0.28)	285.15
		CO_2	5.02		(0.45)	(0.01)	4.56
Rapeseed (b)	FO	Market	338.20				338.20
		CO_2	2.92				2.92
Willow	GRAN	Market	(0.77)		(0.15)		(0.92)
		CO_2			(0.01)		(0.01)
Contaminated							
Energy maize	F + COM	Market	(81.60)	(9.67)	(11.51)	(0.46)	(103.24)
		CO_2	33.77		(0.44)	(0.01)	33.32
Rapeseed (a)	FO	Market	324.06		(38.63)	(0.28)	285.15
		CO_2	5.02		(0.45)	(0.01)	4.56
Rapeseed (b)	FO	Market	338.20				338.20
		CO_2	2.92				2.92
Willow	LF	Market	(3.22)-(1.23)		(0.15)		(3.37)-(1.38)
		CO_2			(0.01)		(0.01)
Marginal impact							
Energy maize		Market					(217.32)
		CO_2					11.18
Rapeseed (a)		Market					0
		CO_2					0
Rapeseed (b)		Market					0
		CO_2					0
Willow		Market					(2.45)-(0.46)
		CO_2					0

COM=combustion, LF=landfill, F=fertilizer, FO=fodder, GRAN=granulates; †D_a is disposal of ashes, T_b is transport of rest products, T_a is transport of ashes, T_g is transport of glycerin; ‡(a) is cake after biodiesel, (b) after PPO

When digestate is uncontaminated the economically most viable option is maximum use as fertilizer and combustion of the rest. This results in a positive economic value of € 114 ha^{-1} $year^{-1}$. When the same digestate comes from metal enriched energy maize, less digestate can be applied on the land as fertilizer and the rest is combusted. This results in a negative economic value of

€ 103 ha^{-1} $year^{-1}$. Therefore, the marginal effect of the metals on the value of the rest products based on valuation only through market prices is € 217 ha^{-1} $year^{-1}$. The same reasoning can be followed for valuation based on CO_2 abatement. Here the marginal impact is positive (€ 33 - € 22). This is because the digestate replaces cokes in combustion without efficiency loss (and this saves more CO_2 than avoiding the production of fertilizer in the uncontaminated scenario). Remarkable but theoretically possible, the presence of metals should not have an economic effect on the economic value of the rapeseed cake. This conclusion is based on current legislation on maximum allowed concentrations in fodder. For willow too, the economic effect of metals on the economic value of the ashes is negligible. This is because the amount of ashes per hectare is small and the economic value of uncontaminated ashes to be used as granulates is also negligible. Therefore, the presence of metals resulting in disposal of the ashes has almost no marginal effect.

Turner (2000) suggests in his paper on waste management that we should take into account external costs and benefits of waste disposal and he criticizes the obliged order of the waste management hierarchy as defined in the European Union (and the Ladder of Lansink in Flanders for that matter). Rather, we should approach waste management with a Cost-Benefit Analysis (preferably accompanied by a Life Cycle Analysis) and choose the option that generates the largest social benefit instead of following the hierarchy blindfolded. This is because properly designed and implemented economic incentive instruments allow any desired level of pollution cleanup to be realized at the lowest overall cost to society. They provide incentives for agents who are able to reduce pollution at reduced costs to contribute more to the final goal than other agents (Hahn and Stavins, 1992). Rather than equalizing pollution levels, economic incentive based approaches equalize marginal abatement costs. Overall efficiency is then achieved when the marginal cost of control is equal to the marginal damage caused by the pollution.

Chapter 3.4 *Energy production and carbon dioxide abatement of crops for sustainable soil management*

This chapter has been submitted in:
Witters, N., Van Slycken, S., Weyens, N., Thewys, T., Meers, E., Tack, F., Vangronsveld, J. (xxxx) Does phytoremediation redeem to the expectations of being a sustainable remediation technology: a case study I: Energy production and carbon dioxide abatement. Biomass Bioenerg.

Abstract

The purpose of this study is to examine the potential benefit of crops used in plant-based contaminated soil management as a resource for renewable energy production. There is an obvious need for remediation and risk reduction alternatives in Europe which are environmentally sound and protective of human health. It is also now widely recognized that cleanup activities of hazardous waste sites may affect emissions of greenhouse gases. In addition, the European Renewable Energy Directive promotes an increase in renewable energy to 20% by 2020. Our analysis is based on a case study in the Campine region (in Belgium and the Netherlands), where agricultural soils are diffusely contaminated with cadmium, lead, and zinc, and are characterized by a sandy texture and relatively low pH. This entails an enhanced risk for uptake of these metals in crops and leaching to the groundwater. Regional policy therefore prescribes that the soils should be remediated, while at the same time it is desirable to keep the income of the farmers at least constant. However, the area has such a large extent (700 km²) that conventional remediation is not applicable. The cultivation of non-food crops on such land offers the opportunity to come up with an approach that efficiently uses contaminated agricultural land to address all of above-mentioned issues and that can be beneficial for both the farmer and the society. Specifically, using biomass originating from contaminated land for energy production or as a feedstock for (chemical) industry may contribute to the reduction of carbon dioxide (CO_2) emissions. Performing a Life Cycle Analysis (LCA), we examined the energy and CO_2 abatement potential of willow (*Salix* spp), energy maize (*Zea mays*), and

rapeseed (*Brassica napus*) after being grown on contaminated soil. We took into account the marginal impact of the metals in the biomass on the energy conversion efficiency and on the potential use of the biomass and its rest products after conversion. Our analysis shows that digestion of energy maize with combustion of the contaminated digestate shows the best energetic and CO_2 abating perspectives. The replacement of cokes based electricity by willow is more efficient in CO_2 abatement than willow used in a CHP unit, despite lower net energy production in the former option. Willow reaches the same energy production and same CO_2 abatement per hectare per year as energy maize when its relative biomass yield (to energy maize) is respectively 0.64 and 0.44 (compared to 0.3 in the base case).

3.4.1 Introduction

Environmental and socio-economic problems are often interconnected. One might therefore argue for a holistic perspective on the problem, because a too narrow focus on one problem at a time can, at worst, make another problem even more serious, or, at best, prevent taking advantage of potential synergy effects (Berndes *et al*., 2008). As is shown in this part, contaminated soil management combined with biomass production to provide feedstock for the production of various biofuels and -products seems to be a good example of where a holistic perspective could be adopted.

Today, it is widely recognized that (conventional) cleanup activities of hazardous waste sites may be the cause of external effects such as the emission of greenhouse gases by the use of heavy duty construction equipment powered by diesel fuel (EPA, 2008). Therefore, in August 2009, the United States Environmental Protection Agency published its proposal called Superfund Green Remediation Strategy which outlines strategic recommendations for cleaner site redevelopment (EPA, 2010). The strategy includes a series of initiatives to stimulate green remediation. The five core elements of green remediation are (i) energy, (ii) air and atmosphere, (iii) water, (iv) land and ecosystems, and (v) materials and waste. Phytoremediation might be able to deal with all these topics. More specifically, in this study, we analyze its potential to use renewable energy sources (and even be a net producer of renewable energy), reduce emissions of greenhouse gases to the air, minimize further harm to the area (biological activity in soil remains untouched), and proper use of the material used for remediation (the biomass, and its rest products after conversion). On a European level, the impact assessment of the Thematic Strategy on Soil Protection summarizes different positive and negative environmental, economic, and social impacts of soil remediation. It mentions the use of energy for the excavation, transport and treatment of contaminated soil as a negative environmental impact of remediation. The European document does not suggest using an alternative technology which might actually result in a net energy production while remediating (COM(2006)231; SEC(2006)1165; SEC(2006)0620).

Within the context of the Kyoto protocol the EU committed to an 8% reduction in CO_2 equivalent[75] emissions in 2008-2012 compared to 1990 levels (for Belgium the proposed reduction is 7.5%) (UN, 1998). Moreover, in 2007, the EU set a series of climate and energy targets to be met by 2020, including a reduction of greenhouse gas emissions by 20% as compared to 1990, the promotion of renewable energy, and an increase of its share to 20% by 2020 (COM(2008)030). On 23 January 2008 the Commission proposed a whole package of binding legislation on the 20-20-20[76] targets, amongst which the Directive 2009/28/EC, the Renewable Energy (RE) Directive (2009) which promotes the increase in renewable energy to 20% by 2020 and the increase in share of biofuels in transport to 10%[77]. For Belgium, the renewable energy objective for 2020 is 13%.

[75] CO_2 equivalents include CO_2 (global warming potential over 100 years (GWP)=1), N_2O (GWP=296), CH_4 (GWP=23), F-gases (GWP>1,000) (www.ipcc.ch).
[76] In its press release of 23.04.2009 (IP-09-628) the Commission states that the accepted energy package will also help achieve the EU's objective of improving energy efficiency by 20% within the same timeframe.
[77] The reasoning behind the 10% biofuels goal can be found in Commission Staff Working Document SEC(2006)1719 based on an impact assessment (COM(2006)0848).

Societies can respond to climate change by adapting to its impacts (adaptation) or by reducing greenhouse gas emissions (mitigation). Specific for our study, on the one hand, the agricultural sector could adapt through rotation variety, while the energy sector could adapt through improved energy efficiency and use of renewable energy sources. On the other hand, the agricultural sector could mitigate through manure management, dedicated energy crops to replace fossil fuel use and improved energy efficiency, while the energy sector could mitigate through fuel switching from coal to gas, renewable heat and power (hydropower, solar, wind, geothermal and biomass), combined heat and power … (IPCC, 2007). In the agricultural sector the focus should not necessarily lay on a lesser use of energy, as this is a negligible part in total energy consumption and emissions in Flanders (Table 3-61 and Table 3-62). We should rather focus on the production and use of renewable energy as a substitute for fossil energy. If this renewable energy cannot be used in the agricultural sector, it might be produced for other sectors.

Table 3-61 Energy use and greenhouse gas emissions of the agricultural sector and its subsectors in 2005, compared to the total in Flanders

Sector	*Energy use in 2005 (PJ)*	*CO_2 emissions in 2005 (kton)*	*CH_4 emissions in 2005 (kton)*	*N_2O emissions in 2005 (kton)*
Agriculture	2.4	902		1,582
Glasshouse	21	1,364		
Intensive cattle	4.2	797	3,635	445
Fisheries	2.5			
Other	2.5			400
Total agriculture	32.7	3,146	3,635	2,440
Total Flanders	2,003	74,692	4,922	5,441

Source: MIRA (2008a, b)

Table 3-62 Energy use as used by different energy carriers in 2005

	Energy use in 2005 (PJ)			
Energy carrier	*Total Agricultural Sector*	*Agriculture*	*Intensive cattle*	*Glasshouse*
Electricity	3.2		1.3	1.4
Heavy oil	9.7			9.7
Diesel	11.9	2.4	2.9	2.1
Gas	6.6			6.6
Cokes	0.8			0.8
LPG	0.5			0.5
Total energy use	32.7	2.4	4.2	21

Source: MIRA (2008a)

Interest in using biofuels as an alternative energy source is high on the agenda of policy makers in many countries. As mentioned by Firbank (2008), a factor that drives the growth of alternative crops for energy is the fact that they deliver an environmental benefit by reducing greenhouse gas (GHG) emissions. However, at the same time, issues rise concerning the fact that these energy crops are often grown on land previously destined for food production. The additional demand for energy during the next years is likely to lead to further increases in energy crop production. Therefore, one way of avoiding the conflict between food and energy crops is to produce biofuel feedstocks on marginal land that is not suited to grow food and fodder crops. This marginal land comprises soils that either, lack nutrients, receive little rain, or have been contaminated due to previous industrial or agricultural activities (Weyens et *al.*, 2009a).

An overview of advantages and disadvantages of conventional remediation technologies and phytoremediation can be found in Witters *et al.* (2009). The current study builds further on the energetic potential generated by crops used for plant-based technologies on contaminated land, as opposed to the energy needed for conventional remediation. We base calculations on samples of energy maize (*Zea mays*), rapeseed (*Brassica napus*), and willow (*Salix* spp.) coming from a moderately contaminated experimental site (6 ha) located in Flanders, Belgium (51°12'41"N; 5°14'32"E) which is part of a larger complex of field experiments for phytoremediation research (~ 10 ha) (Ruttens *et al.*, 2008). From each plant compartment of the different crops fresh and dry weight were measured, and metal content was determined.

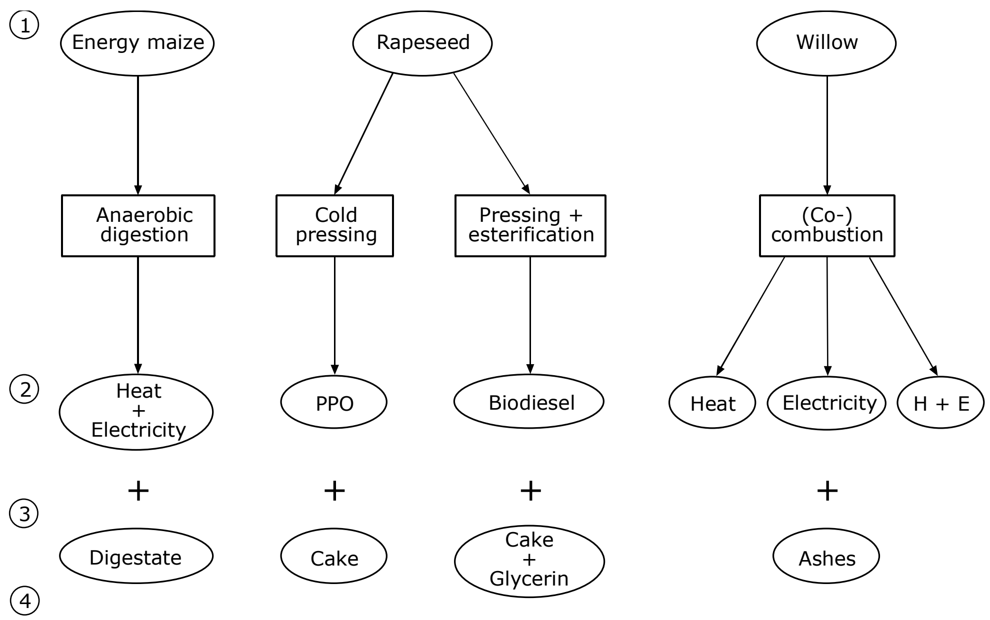
Our study expands the perspective beyond soil management by including the environmental effects that are avoided (CO_2 abatement) during this management. More specifically, the purpose of this study is to examine the potential external benefit of CO_2 abatement resulting from plant-based technologies on contaminated land with energy crops, which would contribute to its label of a sustainable remediation technology. In literature it has been suggested that the utilization of the obtained biomass of a phytoextraction cycle as an energy resource is attractive (Chaney *et al.*, 1997; Dornburg and Faaij, 2005; Licht and Isebrands, 2005; Mirck *et al.*, 2005; Zalesny *et al.*, 2009) and can even turn phytoextraction into a profit making operation (Meers *et al*., 2005a). This study answers the question which crop is best capable of delivering a net benefit to society.

Moreover, when using contaminant enriched biomass crops for energy purposes, the impact of metals on conversion efficiency, as well as the energy needed to properly use or dispose the rest product after conversion should be considered. This has not yet been calculated extensively. Goor *et al.* (2001, 2003) presented a Geographic Information System (GIS) based methodology to evaluate the production of short rotation coppice (SRC) and its conversion into energy on a contaminated district close to Chernobyl. Their model was able to predict biomass production based on soil conditions and thus to present a general overview of potential energy production. However, the model did not proceed to practical implementation and effects of contamination on the model results until Vandenhove *et al.* (2002) built further on the model. They calculated that the contamination scenario would not hamper the economic viability of the energy production schemes, but that feasibility depends on several factors, such as support and capital grants. Our approach differs from theirs in that their analysis is focused on radioactive Caesium (Cs), on SRC of willow, on the extra cost of compensating workforce for the dose occurred, and within a Belarus situation where costs are completely different from those in Western Europe. Our study compares three alternative energy crops grown on Cd contaminated soil in Belgium. The alternative crops were evaluated on energetic and economic grounds, of which only the energetic evaluation will be treated in detail in this chapter performing a Life Cycle Analysis (LCA). Within this energetic analysis we include the impact of Cd on the energetic conversion efficiency of the harvested crop and on further processing and use of the rest product.

3.4.2 Data and methods

In Europe, the production of energy maize is increasing rapidly. The biomass resulting from this crop can be applied for conversion into biogas through anaerobic digestion. As such, energy maize and biogas production represent a new branch of agriculture, which has been emerging at large-scale over the past five to ten years (Meers *et al.*, 2010). Calculations on the energy potential of willow in Belgium seem promising (Cidad *et al*., 2003). After harvest, rapeseed results in rich oil containing seeds, and straw. The use of rapeseed as an income generating crop in Belgium occupies almost 11,000 ha in 2007, most of it being grown in the Southern part of Belgium (FOD Economy: SMEs, independent Professions and Energy, personal communication, March 2009).

To find out whether CO_2 abatement by converting these crops to energy is positive we performed a Life Cycle Analysis (LCA). The International Standards Organization (ISO) defines life cycle assessment as follows: *"A life cycle assessment (LCA) is the assessment of the environmental impact of a given product throughout its lifespan. The comprehensive view provided by LCA is important to avoid system sub-optimisation."* The primary goal of LCA comprises the comparison of the environmental performance of products in order to choose the least burdensome. The term "life cycle" refers to the raw material production, the manufacturing process, the distribution/transport, the final use, and disposal of the product (www.iso.org, 14040 and 14044). LCA attempts to cover all physical exchanges of a product with its surroundings, ranging from inputs of auxiliary materials and energy consumption through outputs of emissions, waste and usable energy (Hanegraaf *et al.*, 1998; Skovgaard, 2008). Figure 3-3 gives an overview of the different steps in determining the potential benefit of CO_2 abatement when using the harvested biomass after remediation for energy purposes.



1 EI = direct+indirect energy input for crop cultivation, including transport to conversion installation

2 EO = total energy output (corrected for conversion efficiencies and process use)

3 $E'_{post\ in\ UC}$ = energy input for the economically most viable end-use of rest product

4 $E'_{post\ out\ UC}$ = energy output for the economically most viable end-use of rest product

Figure 3-3 Biomass conversion routes for alternative crops with indication of CO_2 abatement locations (1-4)

In order to keep the LCA task manageable, a general rule is to focus on obtaining good data on those activities considered most important for the final LCA results (Hanegraaf *et al.*, 1998; Refsgaard *et al.*, 2002; Skovgaard, 2008)[78]. Within the scope of this study it is not a priority to study an entire life cycle, it is therefore more effective to set clear, narrow borders than to set vague borders and include some energy aspects and some not.

[78] Wesseler (2007) disputes the use of indirect energy as he states this brings upon an infinite accounting sequence and an infinite amount of energy used. Depending on the chosen impact category within the predefined borders, results will differ.

3.4.2.1 Assumptions

The studied impact category in our analysis is limited to the Global Warming Potential (GWP) of CO_2. We will not include other greenhouse gases such as CH_4 and N_2O, nor will we discuss ozone depletion, eutrophication (PO_4), or acidification (SO_2). The interested reader can find an overview of the impact of energy crops on these aspects in Braschkat *et al.* (2003)[79]. We do not include a life cycle analysis of the manufacture of the conversion installations, nor the effects of decentralized electricity production on general electricity distribution, for several reasons. First, the main focus of this study is the evaluation of energy crops (grown on contaminated land) and not the technical specifications of the installations. Second, we start from an existing installation, we are not considering a new installation for the conversion of contaminated biomass to energy. And third, we are convinced that specifically designed tools are much better suited for this, see for example the tools developed within the framework of Task 38 of the International Energy Agency on Greenhouse Gas Balances of Biomass and Bioenergy systems (http://www.ieabioenergy-task38.org). We are also aware that the combustion technology needs to be developed and adapted properly to deal with other biomass such as dried digestate as it may lead to excessive corrosion in boiler tubes, excessive slagging and fouling, and higher emissions of NO_x and particulate matter (www.ieabcc.nl). However, no data were available for the case study on this issue yet. While this study takes into account all transportation steps of biomass and rest products, it does not take into account the manufacture of the vehicles used for transportation, nor the manufacture of the vehicles that will use the biofuel. The approach used in this study and suggested by Schlamadinger *et al.* (1997) compares greenhouse gas emissions that arise over the life cycle of each potential technology with those that would have arisen in the fossil situation, allowing for the reduction in emissions to be calculated.

[79]In most cases, biofuels exhibit disadvantages with respect to acidification and eutrophication (Braschkat *et al*., 2003).

Table 3-63 Symbols used in (Eq. 10)-(Eq. 18)

Symbol	*Explanation*	*Unit*
BP_W	potential biomass production of willow	ton dm ha^{-1} y^{-1}
CAL_W	calorific value of willow	MJ ton^{-1} dm
α	loss of biomass/energy due to drying in open air/conditioned drying	%
BP_{EM}	potential biomass production of energy maize	ton fm ha^{-1} y^{-1}
G_{EM}	biogas yield of energy maize	m^3 gas ton^{-1} fm
EV_{BG}	energy value of biogas	MJ m^{-3} gas
BP_{RS}	potential biomass production of rapeseed	ton fm ha^{-1} y^{-1}
G_{PPO}	efficiency of mechanical rapeseed pressing to PPO	ton oil ton^{-1} fm
D_{PPO}	density of PPO	ton oil l^{-1} oil
EV_{PPO}	energy value of PPO	MJ l^{-1} oil
G_{BD}	efficiency of rapeseed conversion to biodiesel	ton fuel ton^{-1} fm
D_{BD}	density of biodiesel	ton l^{-1}
EV_{BD}	energy value of biodiesel	MJ l^{-1}
EI	total primary energy input	MJ ha^{-1} y^{-1}
EO	total energy output	MJ ha^{-1} y^{-1}
EI_{pred}	direct energy input crop cultivation, including transport crop to installation	MJ ha^{-1} y^{-1}
EI_{prei}	indirect energy input during establishment of the crop	MJ ha^{-1} y^{-1}
GEC	gross energy content	MJ ha^{-1} y^{-1}
η(th),η(el), η(m)	conversion efficiency (resp. thermal, electric, and mechanical)	%
β(th),β(el), β(f)	fossil energy use during conversion (resp. thermal, electric and diesel fuel)	%
E_{post}	output ($E_{post\ out}$) - input ($E_{post\ in}$) of energy for secondary use rest product	MJ ha^{-1} y^{-1}

3.4.2.2 Fossil energy input for biomass production

Direct energy use refers to the fossil fuel consumed within the borders of the farm. Indirect energy use refers to fuel burned in other sectors that manufacture the materials needed at the farm. Following the example of Refsgaard *et al.* (2002), we set the boundaries one step back from the farming process (Table 3-64). Adding direct and indirect energy input results in total fossil energy inputs (Table 3-70, Table 3-71, and Table 3-72).

Table 3-64 Direct and indirect energy input for biomass production

Direct	*Indirect*
-Diesel fuel use machines (plowing, disking, planting, cultivation, application of fertilizers, herbicides, lime and manure, and harvest), corrected for extraction and distribution	-Production and transport fertilizer, lime, and pesticides (herbicides, insecticides, and fungicides)
	-Production of seeds
	-Manufacturing, transport and reparation machines
-Use of lubricants for machines	
-Irrigation	
-Ensiling	
-Drying	
-Transport crops to conversion installation (Table 3-8)	

3.4.2.3 From biomass to gross energy

Table 3-4 provides an overview of all crop conversion options analyzed in this study. Symbols used in the formulae can be found in Table 3-63, values in Table 3-65.

Energy maize is digested anaerobically, a conversion process where organic matter of biomass is converted into methane in four phases by bacteria in the absence of oxygen. The end products of the digestion process are biogas and digestate. Due to its high energy content, biogas can be used in engines and machines to replace natural gas, be used as a transport fuel, or even be injected in the natural gas distribution network[80] (Verstraete, 1981; Ramage and Scurlock, 1996). We opted for the first choice, burning the gas in a gas engine for the production of electricity with heat recovery in a combined heat and power (CHP) system. Cogeneration can be defined as the thermodynamic sequential production of two or more energy forms starting from only one primary energy

[80]The quality of the biogas has to be elevated to the level of natural gas prior to injection in the gas network. Levels of CO_2, H_20, H_2S and NH_4 have to be brought to a lower level (Senternovem, 2006; www.energietech.info/groengas/theorie/opwerken.htm).

source (Cogen Vlaanderen vzw, 2006). These energy forms are very often thermal and mechanical energy, where the latter is used to produce electricity, hence the often used name Combined Heat and Power (CHP). Electric and thermal efficiency of a CHP are lower than for the separate production of electricity and heat, but the simultaneous production of electricity and heat from one input renders a higher overall energetic efficiency. The gross energy content of energy maize (per ha) after digestion (GEC_{EM}) is calculated in (Eq. 10):

$$GEC_{EM} = BP_{EM} \cdot G_{EM} \cdot EV_{BG} \qquad \text{(Eq. 10)}$$

Mechanical pressing of rapeseed can be warm or cold. The resulting products are pure plant oil (PPO), and rapeseed cake (cold) or scrap (warm). Cake is a marketable co-product, used mainly as cattle feed. Warm pressing renders a higher percentage of available oil (32-42 m%) than cold pressing (28-35 m%). On a large scale (*e.g.* prior to biodiesel production), chemical pressing renders 42 m% of PPO. Further transesterification of the PPO results in biodiesel. Biodiesel can be burned in a regular diesel engine as such to replace fossil diesel, whereas for the combustion of PPO, some adjustments to the engine are necessary. After transesterification of PPO, crude glycerin is obtained (Eq. 11) (Van de Plas, 2007).

$$1 \text{ ton PPO} + 0.1 \text{ ton methanol} \rightarrow 1 \text{ ton biodiesel} + 0.1 \text{ ton glycerin} \qquad \text{(Eq. 11)}$$

The gross energy content of PPO and biodiesel per hectare (GEC_{PPO} and GEC_{BD}) are calculated in (Eq. 12) and (Eq. 13) respectively.

$$GEC_{PPO} = BP_{RS} \cdot (G_{PPO} / D_{PPO}) \cdot EV_{PPO} \qquad \text{(Eq. 12)}$$

$$GEC_{BD} = BP_{RS} \cdot (G_{BD} / D_{BD}) \cdot EV_{BD} \qquad \text{(Eq. 13)}$$

Combustion, defined by Demirbas (2003), involves oxidation of biomass with excess air, providing hot flue gases that are used to produce steam in the heat exchange sections of the boiler. For the production of electricity, the produced steam is expanded under high pressure in a steam turbine. Co-combustion is defined as the combustion of a renewable fuel (biomass) along with the primary fuel (natural gas, coal,…).

For SRC of willow, we consider three options. First, co-combustion of biomass with coal has already been widely applied (Goovaerts *et al.*, 2009) and its economic potential has been indicated (Hughes, 2000). Willow from phytoremediation could be co-combusted in a power plant replacing coal (Electrabel). Currently, this installation (540 MWe) indirectly co-combusts pulverised coal with woodchips from recycled fresh wood and hard and soft board (www.ieabcc.nl). We consider another installation as this one has a $deNO_x$ and $deSO_x$ installation. The maximum yearly potential for co-combustion would be 200,000 ton (5%). Nussbaumer and Oser (2004) conclude that it can be assumed that the energy needed for the pre-treatment of wood chips is included in the efficiency of power production. Theunis *et al.* (2003a) also mention that coal cannot be used as such in a coal power plant but has to undergo some pre-treatment which results in coal powder. The same pre-treatment is needed for the wood chips that replace coal. Second, willow could be used for heating purposes to replace cokes in the Zinc smelter in the Campine region. Since the startup of the copper smelter in 1997, first as part of the smelter, and from 2007 on as an independent company, secondary sources are co-combusted, according to the license. To result in the same heat output, relatively more biomass has to be burned than coal due to the lower heating value of the former. Since this requires additional storage, handling, and transport, there is a maximum amount of biomass that can be co-fired for the installation to still be economically viable (Sami *et al.*, 2001; Baxter, 2005; Eriksson, 2007). Third, we consider the combustion of willow in an existing biomass based combustion installation with electricity and heat production in a CHP system. This replaces the separate production of heat (natural gas) and electricity (average fossil mix). The gross energy content of SRC is based on its calorific value, which is the amount of energy present in the wood and liberated when burned. The gross energy content of willow (after drying) per hectare (GEC_W) is calculated in (Eq. 14).

$$GEC_W = BP_W \cdot CAL_W \cdot (1-\alpha) \qquad \text{(Eq. 14)}$$

Table 3-65 Base case values used in (Eq. 10)-(Eq. 18)

Symbol	*Base case*	*Unit*
(Eq. 10)-(Eq. 14)		
BP_W	6†	ton dm ha^{-1} y^{-1}
CAL_W	19.92‡	GJ ton^{-1} dm
α	0	%
BP_{EM}	60†	ton fm ha^{-1} y^{-1}
G_{EM}	190§	m^3 gas ton^{-1} fm
EV_{BG}	19.11¶	MJ m^{-3} gas
BP_{RS}	3.3†	ton fm ha^{-1} y^{-1}
G_{PPO}	35%#	ton oil ton^{-1} fm
D_{PPO}	0.90††	ton l^{-1}
EV_{PPO}	34.11††‡‡	MJ l^{-1}
G_{BD}	42%#	ton biodiesel ton^{-1} fm
D_{BD}	0.88‡‡	ton l^{-1}
EV_{BD}	33.18‡‡	MJ l^{-1}
(Eq. 15)-(Eq. 18)§§		
Energy value diesel	35.9	MJ l^{-1} diesel
Extraction and distribution diesel	5	MJ l^{-1} diesel
Lubricants	3.6	MJ l^{-1} diesel
Energy use transport	1.3	MJ ton^{-1} fm km^{-1}
Electr. use separating digestate (11 dm% to 30 dm%)	27-36	MJ m^{-3} input
Heat use drying digestate (30 dm% to 85 dm%)	2.37	GJ ton^{-1} separated digestate
Energy value coal/cokes	29	GJ ton^{-1} cokes

†Based on Table 3-1, Table 3-2, and Table 3-3; ‡Vande Walle (2007b); §Thewys *et al.* (2010a, b); ¶53.25% CH_4, Lower Heating Value LHV(CH_4) = 35.9 MJ m^{-3}; #GAVE (2005); ††www.hanze.nl, www.wervel.be; ‡‡www.emis.vito.be, RE Directive (2009/28/EG); §§Borjesson (1996), Dalgaard *et al*. (2001), Lemmens *et al.* (2007), www.emis.vito.be

3.4.2.4 Effect of metals on energy production potential

Metal concentrations found in biomass from the Campine region can be found in Table 3-1, Table 3-2, and Table 3-3. We assumed for the three studied crops that metals have no marginal effect on the energy conversion (technical) efficiency (0). Moreover, no metals end up in the energy carrier, but are concentrated in the rest products: ashes, digestate, and cake. In this part we can thus focus on energy production and CO_2 abatement for each of the crops as if there were no metals present.

3.4.2.5 From gross energy content to net thermal, electric, and mechanical energy

The CHP reference Decision (2006) defines electric efficiency as net electricity production divided by total fuel use, expressed by its lower heating value (LHV)[81]. Thermal efficiency is defined as net heat divided by total fuel use, expressed by its LHV.

The electric efficiency of the combustion of woody biomass in a coal power plant (37%) is based on the reference efficiency of 33% as defined in the CHP reference Decision (2006), and consistent with Theunis *et al.* (2003a). Co-firing power plants have better electric efficiency compared to 100% biomass based power plants, but compared to coal there is a 0-10% efficiency loss in biomass conversion, due to use of non-preheated air (which could be avoided in a permanent installation), energy use for preparation and handling of the biomass, and a higher moisture content of the biomass (Baxter, 2005). The second option is combusting willow for heating purposes to replace cokes in the Zinc smelter. When 7-10% of biomass is co-fired, there is a drop in overall boiler efficiency compared to coal-fired boilers. This reduction is however minimal, 0.3-1.0 points of the 85-90% thermal efficiency (Hughes, 2000). As thermal efficiency for combusting woody biomass we use 78% (CHP reference Decision, 2006). Finally, wood could also be combusted with other biomass in a biomass (waste) incineration plant for the generation of heat and electricity using a steam turbine based CHP. Examples have been studied by the International Energy Agency for Biomass Combustion and Co-firing (IEABCC, 2004) in Denmark and Austria. Steam turbines have a low electric efficiency (Cogen Vlaanderen vzw, 2006).

[81]LHV is the total amount of heat delivered by complete combustion of a fuel, without condensation of water vapor in the combustion gases.

Thermal and electric efficiency (69% and 16%) of this system are based on Vande Walle *et al.* (2007b).

The produced gas resulting from digesting energy maize is burned in a gas engine based CHP. This replaces the separate production of heat and electricity with fossil fuels. The thermal and electric efficiency of a gas engine are 45% and 40% respectively (Table 3-66).

Table 3-66 Relevant conversion efficiencies to be applied on the GEC or CAL of the different options (η=efficiency, β=use, el=electricity, th=thermal, f=fuel)

Conversion technology	*Net energy*	*η(th)*	*η(el)*	*β(th)* †	*β(el)* †	*β(f)*
Co-combustion	Electricity		37%[¶ #]		4%[#]	
Co-combustion	Heat	78-80%[¶ #]				
Combustion	CHP (steam turbine)	55-69%[# ††]	16%[# ††]	4%[‡]	3%[‡]	
Digestion	CHP (gas engine)	45%[‡‡]	40%[‡‡]	30%[§§]	7.5%[§]	
Pressing	PPO				2%[¶¶]	
Transesterific.	Biodiesel				2%[¶¶]	8%[¶¶]

[†]% of CAL_W, % of GEC_{EM}, and % of GEC_{RS}; [‡]Cidad *et al.* (2003); [§]Goossens (2007); [¶]CHP Ref Decision (2006); [#]Theunis *et al.* (2003a); [††]Vande Walle *et al.* (2007b), Cogen Vlaanderen vzw (2006), IEABCC (2004); [‡‡]Stroobandt, A., personal communication (January 2009); [§§]Meers, E., personal communication (March 2007); [¶¶]Gustavsson *et al.* (1995), Janulis (2004), and 8% is assumed for the production of chemicals

3.4.2.6 Metal enriched rest product

After conversion, a rest product remains. Besides generating energy output ($E_{postout}$), the rest product needs energy input prior to secondary use (E_{postin}). For all rest products, E_{postin} also includes transport (Table 3-67). We have not included the energy cost of applying digestate (0.6 l diesel ton^{-1}) since we assume that this is compensated by applying less chemical fertilizer (2 l ha^{-1}) (Cidad *et al.*, 2003). Moreover, we did not take into account the energy cost of upgrading the ashes to granulates and will also not take into account the energy cost of the replaced building materials by these granulates (Table 3-67).

Table 3-67 Overview of marginal energy in- and output for secondary use of rest products

Rest product	$E_{post\ in}$	$E_{post\ out}$
Digestate (11 dm%)	Separation (30 dm%) + drying (85 dm%) T(digestate to field) T(digestate to combustion install) T(ashes to landfill)	Energy cost chemical fertilizer (including T to farm) Combustion value digestate (heat)
Cake	T(fodder to farm)	Energy cost roughage production (including T to farm)
Glycerin	T(glycerin to digester)	Digestion
Ashes	T(ashes to landfill/granulate use)	/

T=transport

The presence of metals may have an effect on secondary use options of the rest product, and this effect is economically valued in Chapter 3.3 through (i) the marginal impact on the market value of the rest product (due to restricted options for secondary use), followed by (ii) the marginal impact on final CO_2 abatement. By comparing the end use of the rest products in the uncontaminated scenario with the contaminated scenario (Table 3-59), we were able to economically value the impact of metals in the rest product.

Therefore, to avoid double counting, we base calculations on our energy budget and CO_2 abatement on uncontaminated rest products.

Uncontaminated digestate is a good alternative for chemical fertilizer or could be used as a soil amendment. Due to N, P, and K threshold values, not all digestate (47 ton) resulting from one hectare of energy maize can be applied on that same hectare. Approximately 33 ton uncontaminated digestate could be applied per hectare. Only 13.77 ton uncontaminated (11 dm%) digestate is separated and dried resulting in 1.78 ton (85 dm%) for combustion, resulting in 0.07 ton ashes which will be landfilled. In the contaminated best option, 44.4 ton digestate (11 dm%) should be separated and dried resulting in 5.76 ton (85 dm%) for combustion. This is because besides the restrictions on N, P, and K, there are also restrictions regarding the maximum allowed trace element concentrations in the digestate if it is to be used as fertilizer or soil improving material. Additionally, combustion of digestate results in 0.23 ton ashes that will be landfilled.

1 ha rapeseed (3.3 ton fm) generates respectively 1.17 and 1.40 ton oil (in PPO and biodiesel scenario), and respectively 2.16 and 1.93 ton cake. In the biodiesel scenario, the oil is subsequently converted to biodiesel through transesterification (100%): 1.40 ton rapeseed oil + 0.14 ton methanol → 1.40 ton biodiesel + 0.14 ton glycerin. By comparing legislation and concentrations of metals found in rapeseed cake, we ascertained that this product can still be used as fodder. Rapeseed cake does not need to undergo any energy consuming pre-treatment before use as fodder.

Willow (co-) combustion results in ashes which can be used as granulates according to current legislation. When using uncontaminated woody biomass, no pre-treatment is necessary. When the woody biomass is contaminated, the ashes need to be landfilled.

In summary, we found the following effect of metals on end use of the rest product in Chapter 3.3. For energy maize there was a private economic effect due a different fertilizer/combustion ratio of the digestate. This also resulted in a different energy budget and CO_2 abatement (Table 3-68). Because of metals in the rest product, $E_{post\ in}$ for contaminated digestate is 40.55 GJ ha^{-1}. $E_{post\ out}$ is 53.01 GJ ha^{-1}. This results in E_{post} = 12.46 GJ ha^{-1}, implying a net positive impact of metals on energy production of 250 MJ ha^{-1} compared to the uncontaminated scenario. The presence of metals forces us to use the digestate for other purposes which are more energy efficient than its use in the reference uncontaminated situation. This also has an effect on CO_2 abatement, compared to the reference uncontaminated scenario. The effect is completely due to alternative use of the rest product, where in the uncontaminated and contaminated scenario the net CO_2 avoidance are respectively (2,815-1,729) kg CO_2 and (7,054-5,375) kg CO_2. The contaminated scenario needs heat to dry the digestate, but combustion generates heat (replacing cokes), whereas in the uncontaminated scenario, less energy is necessary, since most of the digestate is applied on the field, but the avoided energy for chemical fertilizer results in less CO_2 avoidance (produced with a combination of cokes, gas, and electricity).

For rapeseed we concluded that there was no economic effect of metals on use of the rest product. For willow, there was only a private economic effect due to the different cost of disposal and use for granulates. There was no impact of metals in ashes on $E_{post\ in}$, and $E_{post\ out}$ since we assume that transport distances are the same for uncontaminated and contaminated ashes. Moreover, we did not take into account the energy cost of replaced building materials by granulates in the uncontaminated scenario.

Table 3-68 Energy input (Epostin) and output (Epostout) of digestate (GJ ha^{-1}) and resulting CO_2 abatement (kg ha^{-1}) in uncontaminated (UC) and contaminated (C) scenario

		UC			*C*		
	GJ ton^{-1}	*Ton ha^{-1}*	*GJ ha^{-1}*	*kg CO_2 ha^{-1}*	*Ton ha^{-1}*	*GJ ha^{-1}*	*kg CO_2 ha^{-1}*
Epostin			*13.42*	*5,375.16*		*40.55*	*1,729.20*
separation	0.036	13.77	0.50		44.40	1.60	
drying	0.870	13.77	11.98		44.40	38.65	
transport to combustion	0.039	1.78	0.07		5.76	0.22	
transport to farm	0.026	33.43	0.87		2.80	0.07	
transport ashes	0.039	0.07	0.00		0.23	0.01	
Epostout			*25.63*	*7,053.80*		*53.01*	*2,840.93*
chemical fertilizer	0.282	33.43	9.44		2.80	0.79	
combustion value	1.18	13.77	16.19		44.40	52.22	
Epost			*12.21*	*1,678.64*		*12.46*	*1,111.73*

Based on Chapter 3.3

3.4.2.7 Overview

$EI = EI_{pred} + EI_{prei}$ (Eq. 15)

$EO = GEC \cdot \eta(th) \cdot (1-\beta(th)) + GEC \cdot \eta(el) \cdot (1-\beta(el))$ (Eq. 16)

$EO = GEC \cdot (1-\beta(f) - \beta(el))$ (Eq. 17)

$E_{post} = E_{postout} - E_{postin}$ (Eq. 18)

(Eq. 15) calculates the total (direct and indirect) energy input for willow, energy maize and rapeseed. (Eq. 16) represents the energy output from energy maize and willow, while (Eq. 17) is used for rapeseed. (Eq. 18) calculates the net energy due to secondary use of the rest product. The total primary energy input only contains processes where actual fossil energy is used. Renewable energy uses or losses, such as during natural drying for willow (α) or during conversion to heat and electricity (η and β), will not be considered as energy inputs. They are considered losses of energy output and as such were subtracted from the output. This is different from the approach used by Vande Walle *et al.* (2007b) where the use of electricity and heat during conversion were considered as energy input.

Bioenergy production systems can be compared based on several criteria (Table 3-69). Different energy and carbon budgets can be calculated: net energy (NE), energy ratio (ER), net energy requirement (NER), and gross energy requirement (GER) (Matthews, 2001).

Table 3-69 Energy and carbon budgets

Ratio	*Formula*	
NE	EO-EI	Number of energy units produced by system after fossil energy input has been deducted
ER†	EO / EI	Number of energy units produced by system per unit of fossil energy input to drive system (%)
NER	EI / EO	Input needed to drive the system per unit of energy produced (%)
GER	(EO+EI) / EI	Amount of energy produced after input has been deducted (preferably >1)

†ER (%) is often used as it gives more information about the degree to which a given fuel is or is not renewable. If ER=0, then the produced fuel is completely nonrenewable. If ER=1, then this fuel is still nonrenewable. It only means that no loss of energy occurs in the process of converting the fossil energy to a usable fuel. If ER>1, the produced fuel begins to provide a leveraging of the fossil energy required to make the fuel available (Sheehan *et al.*, 1998).

For each of the energy crops and their conversion routes in Table 3-4, with the most viable options for the rest products in Table 3-59, we calculated the net energy production (NE) per hectare per year[82]. Based on NE (EO + E_{post} – EI), we calculated the net avoided CO_2 emission.

3.4.3 Results

3.4.3.1 Net energy

Energy maize

Energy maize is digested and the resulting biogas is combusted in a gas engine in a CHP system, resulting in heat and electricity (Table 3-70). The digestate is partly used as fertilizer, partly combusted, according to legislation (Table 3-59). Transport costs of energy maize to the digester are based on a 20 km distance. Transport of digestate is 20 km back to the farm, used as a fertilizer, 30 km to Nyrstar (for combustion). After combustion, transport of ashes to a landfill is 30 km.

Table 3-70 Energy costs for crop growth and net energy gain after conversion of energy maize (MJ ha^{-1} year^{-1})

	Energy (MJ ha^{1} year^{-1})	*Source*
Crop growth	6,770	(Cidad *et al.*, 2003; Dalgaard, 2001)†
Transport energy maize to digester	1,560	(Borjesson, 1996)
Indirect energy	5,040	(Cidad *et al.*, 2003)
EI	*13,370*	
GEC_{EM}	217,810	‡
Net electricity	80,590	
Net heat	68,610	
EO	*149,200*	(Cidad *et al.*, 2003; Goossens, 2007)
Epostin	13,419	§
Epostout	25,630	
Epost	*12,211*	
NE	*148,050*	

†152 l diesel ha^{-1}; ‡60 ton fm ha^{-1} y^{-1}, 190 m³ gas ton^{-1} fm, 19.11 MJ m^{-3} gas ; §Table 3-68

[82]Because of the difficulties with interpretation of ratios and because we need an absolute number to internalize the externality of CO_2 abatement by biomass after remediation.

Willow

We use a mean calorific value for willow of 19,920 MJ ton^{-1} dm (Vande Walle *et al.*, 2007b). Table 3-71 gives an overview of total average energy costs for willow per hectare per year, based on an average total yearly dry matter yield of 6 ton per hectare per year. For willow, the largest contributor to the energy cost is the indirect machinery, followed by planting material preparation, and final stool removal. Transport costs of willow are based on a 30 km distance to a (co-) combustion installation.

Table 3-71 Energy costs for crop growth and net energy gain after conversion of willow (MJ ha^{-1} $year^{-1}$) based on biomass production of Belgisch Rood and Jorunn

	Energy (MJ ha^{-1} $year^{-1}$)			*Source*
Crop growth	8,410†			(Cidad *et al.*, 2003; Dalgaard, 2001)
Transport SRC to co-combustion	330			(Borjesson, 1996)
Indirect energy	2,610			(Cidad *et al.*, 2003)
EI	*11,350*			
GEC_W	95,620‡			(Vande Walle *et al.*, 2007b)
	(1)	(2)	(3)	
Net electricity	31,550		12,430	
Net heat		74,580	62,150	
EO	*31,550*	*74,580*	*74,580*	
Epostin	3§	3§	3§	
Epostout	0	0	0	
Epost	*-3*	*-3*	*-3*	
NE	*20,200*	*63,230*	*63,230*	

†189 l diesel ha^{-1} y^{-1} is an average yearly use over 21 years; ‡4.8 ton dm ha^{-1} y^{-1} (excl. leaves) after first rotation cycle (15-20 ton over 3 years), (1) electricity, (2) heat, (3) CHP; §30 km transport distance, 76.8 kg ashes

Rapeseed

Rapeseed results are represented in Table 3-72. The same amount of rapeseed cake can be used as fodder in the uncontaminated and contaminated scenario (Table 3-59).

Table 3-72 Energy costs for crop growth and net energy gain after conversion of rapeseed (1) into PPO and (2) into biodiesel (MJ ha^{-1} $year^{-1}$)

	Energy (MJ ha^{-1} $year^{-1}$)		*Source*
	(1)	*(2)*	
Crop growth	3,130	3,130	(Cidad *et al.*, 2003)
Transport rapeseed to biodiesel/PPO + transport straw	60	580	(Borjesson, 1996)
Indirect energy	10,900	10,900	(Cidad *et al.*, 2003)
EI	*14,090*	*14,540*	
GEC_{PPO}	44,170†		
GEC_{BD}		52,730†	
Electricity use	880	1,050	
Fuel use		4,220	
EO‡	*43,290*	*47,450*	
Epostin§	0	300	
Epostout¶	1,970	3,090	West and Marland (2002)
Epost	*1,970*	*2,790*	
NE	*31,170*	*35,640*	

†3.3 ton rapeseed ha^{-1} y^{-1}; ‡we did not take into account the transport of biodiesel or PPO after conversion; §transport of cake and glycerin after biodiesel; ¶net energy (MJ) saved per ton cake is the energy needed to produce the avoided amount of regular fodder, in the biodiesel scenario, the energy production from digestion of glycerin is added; 750 m³ gas per ton glycerin (Sys, K., personal communication, April 2010) resulting in 720 MJ electricity and 610 MJ heat

3.4.3.2 CO_2 abatement

Table 3-73 CO_2 emission coefficients from combustion of fossil energy carriers (g kWh^{-1})

	Natural gas	*Cokes*	*Coal*	*Average*	*Lubricants*	*Diesel*
100%	201.96†	385.2†			263.88†	266.76†
Engine (η(m))					*659.7 (40%)*‡	*666.9 (40%)*‡
Heat	*224.4 (90%)*§	*481.5 (80%)*§				296.4 (89%)§
Electricity	116-398††		*508-897*††	*413.18*¶		
CHP (η(el))		1,540.8 (25%)#				

†IPCC (2007); ‡Cidad *et al.* (2003); §CHP reference Decision (2006); ¶MIRA (2008b); #Cogen Vlaanderen vzw (2006); ††Born (n.d.), Envirochem (2005)

The yearly CO_2 abatement per hectare for each option can be found in Table 3-74, based on CO_2 emission coefficients for each of the fossil energy carriers in Table 3-73. In calculating CO_2 abatement, the emission of fossil fuels is corrected for engine and boiler efficiency. This is not the case for rapeseed where the production of biodiesel or PPO represent output before conversion in an engine. The extraction of fossil diesel is also taken into account, as well as the use of lubricants. Transport and indirect energy use always assume the use of fossil diesel. During biodiesel production, diesel fuel is used (8% of gross biodiesel production). The heat used for drying digestate comes from heat produced in the digester which takes into account efficiency of the CHP engine. The calculation of CO_2 emission avoidance for digestate used as fertilizer is based on the energy use for the production of chemical fertilizer and includes transport. The calculation of CO_2 abatement by combustion of digestate for heat production is based on a comparison with heat based cokes. Avoided CO_2 emissions by the use of rapeseed cake for fodder are based on diesel use during crop growth of the regular fodder (Chapter 3.3).

Table 3-74 Net CO_2 avoidance per hectare per year for energy maize, SRC of willow, and rapeseed (kg CO_2 ha^{-1} $year^{-1}$) (net emissions between brackets)

CO_2 (cultivation)	*Energy maize*		*SRC*			*Rapeseed*
diesel use	(950)	(791)	(791)	(791)	(1,009)	(1,047)
lubricant use	(40)	(50)	(50)	(50)	(34)	(34)
CO_2 (conversion)(EO)	*Digestion*	*Electr.*	*Heat*	*CHP*	*PPO*	*Biodiesel*
electricity	+9,249	+6,157	0	+1,427		
heat	+4,277	0	+9,975	+3,874		
diesel					+3,273	+3,907
extraction diesel					+456	+544
electricity use					(101)	(121)
fuel use						(313)
net CO_2 avoided	*12,536*	*5,317*	*9,135*	*4,460*	*2,584*	*2,936*
Rest product sec use	*fertil + comb*	*landfill*	*landfill*	*landfill*	*fodder*	*fodder*
Epostin	(1,729.20)	(0.22)	(0.22)	(0.22)	0	(22)
Epostout	+2,840.93				+146	+251
net after rest product (NE)	*13,647.59*	*5,316*	*9,134*	*4,460*	*2,730*	*3,044*

All crops offer the potential to reduce CO_2 emissions (Table 3-74). Digestion of energy maize offers the best potential. SRC of willow to replace cokes based heat (Nyrstar) comes second. Alternatively, SRC of willow in a biomass combustion installation and the combined production of heat and electricity comes only after the separate production of electricity from willow, since in the latter case willow replaces cokes (Electrabel), and in the former it replaces average electricity and heat.

3.4.4 Discussion and conclusion

On a European level, the impact assessment of the thematic strategy on soil protection summarizes different positive and negative environmental, economic, and social impacts of soil remediation. The extensive use of energy for the excavation, transport and treatment of contaminated soil is a negative environmental impact of remediation. Moreover, the destruction of natural structures is mentioned as one of the major environmental drawbacks of conventional remediation. As an economic drawback, the high cost of conventional remediation is mentioned, if not covered by the company causing the pollution (according to the widely used Polluter Pays Principle), then by the public authority (COM(2006)231). In this study we demonstrated the potential of plant-based technologies to deal with the former drawback, but we are of the opinion that plant-based technologies have more to offer and might actually be capable of overcoming other negative issues often met when using conventional remediation technologies as well.

From the life cycle analysis for each of the crops and their conversion technologies (Table 3-74) we conclude that all crops offer the potential to generate net renewable energy, and reduce CO_2 emissions. However, results are highly dependent on the used Flemish/Belgian reference efficiencies and the reference fossil situation. For the decision on reference situations, we opted for the practically achievable ones. For example, combustion of willow in a biomass installation with the combined production of heat and electricity might intuitively seem to have a competitive CO_2 abatement advantage over co-combustion for the production of heat or electricity, but in our case it does not, because we compare with a very bad reference situation with regards to CO_2 emission, since electricity and heat is often still generated by combustion of cokes.

Two aspects are worth deeper investigation:
(i) the change in best resulting crop when focus lies on net energy production versus net CO_2 abatement; and
(ii) the promising perspective of a non-commercial willow clone from the INBO breeding program, which shows promising energy and CO_2 abatement perspectives. The average biomass productivity of 6 ton dm ha^{-1} $year^{-1}$ is low in comparison to the average expected productivity values found in literature (10-12 ton dm ha^{-1} $year^{-1}$). Ongoing trials (on the same site in Lommel) with clones that are not yet available on the market (breeding program of the Research Institute for Nature and Forest, INBO, Belgium) showed that a production of 15.6 ton dm ha^{-1} $year^{-1}$ could be reached (unpublished data). This has an impact on net energy production per hectare and net CO_2 abatement per hectare.

In our study, energy maize scores best on net energy production and CO_2 abatement potential (when we leave the non-commercial willow clone aside). Figure 3-4 represents net energy production versus net CO_2 abatement for each crop conversion option. Drawing a straight line through the origin and this point, reveals how the other crop-conversion pathways are positioned compared to energy maize. Options on the line have the same "net CO_2-net energy" ratio as energy maize, while points below the line have a lower "net CO_2-net energy" ratio, and points above have a higher "net CO_2-net energy" ratio. Figure 3-4 shows that both rapeseed conversion options (PPO and biodiesel) lie on the line, indicating that both options can only become better than energy maize, when moving along the line, and thus by improving energy efficiency, and biomass yield. Willow combined with CHP production lies below this line, implying that the net energy production does not result in as much CO_2 abatement as energy maize. It might therefore be concluded that willow might better be used for other options for which the net energy results in a higher CO_2 abatement. An example is the conversion of willow to replace cokes based electricity generation. Although in our case study, this crop-conversion option shows a rather low net energy production potential, its "net CO_2-net energy" ratio shows promising perspectives concerning its potential contribution to CO_2 abatement. We therefore conclude that, despite its lower net energy production, the replacement of cokes based electricity by willow is a better (more efficient in CO_2 abatement) option than willow used in a CHP unit. This does however not mean that in the long run willow should not be used for combined conversion into heat and electricity, but only after all electricity installations with cokes have been converted into willow (or other biomass) based installations or have been replaced by a CHP turbine.

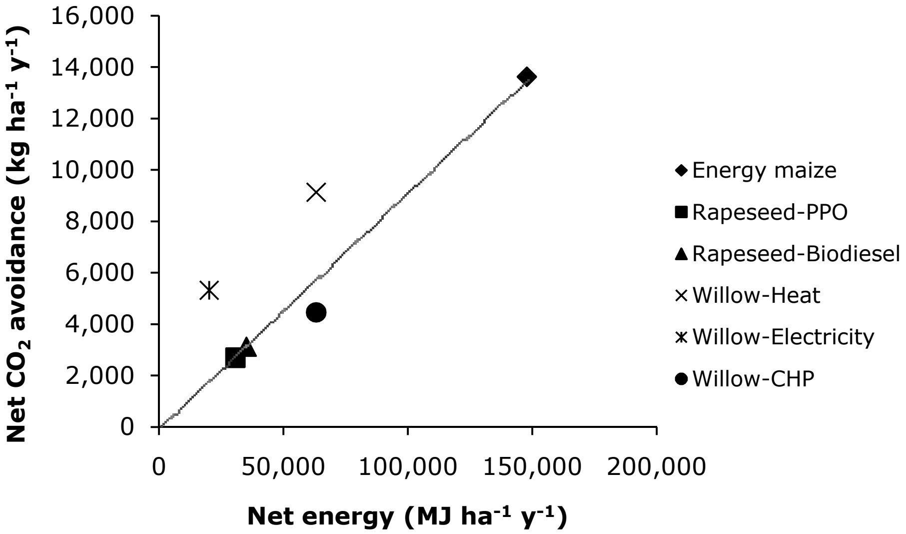


Figure 3-4 Net CO_2 avoidance (kg ha^{-1} year^{-1}) versus Net Energy production (MJ ha^{-1} year^{-1}) for each crop conversion option

In our calculations, we used a ratio in biomass yield (dm) of 0.3 (based on Table 3-1 and Table 3-3), resulting in net energy ratios between 0.14 (co-combustion of willow in a cokes based electricity installation) and 0.44 (co-combustion of willow in a new CHP turbine), and CO_2 abatement ratios between 0.33 (combustion of willow in an existing CHP installation) and 0.67 (co-combuston of willow to replace cokes based heat). Stated otherwise, SRC of willow has a yearly dry biomass yield that lies 70% lower than that of energy maize, which translates in a net energy production that lies 56-86% lower. This shows perspectives for SRC of willow, for its CO_2 abatement which lies only 67-33% lower than that of energy maize. Given this conclusion and the biomass potential of one of the non-commercial willow clones from the INBO breeding program, we included Figure 3-5 where we studied the impact of a change in relative biomass yield (dm) of willow to energy maize on (i) the energy ratio (E) and on (ii) the CO_2 abatement ratio (C) for different willow-conversion options.

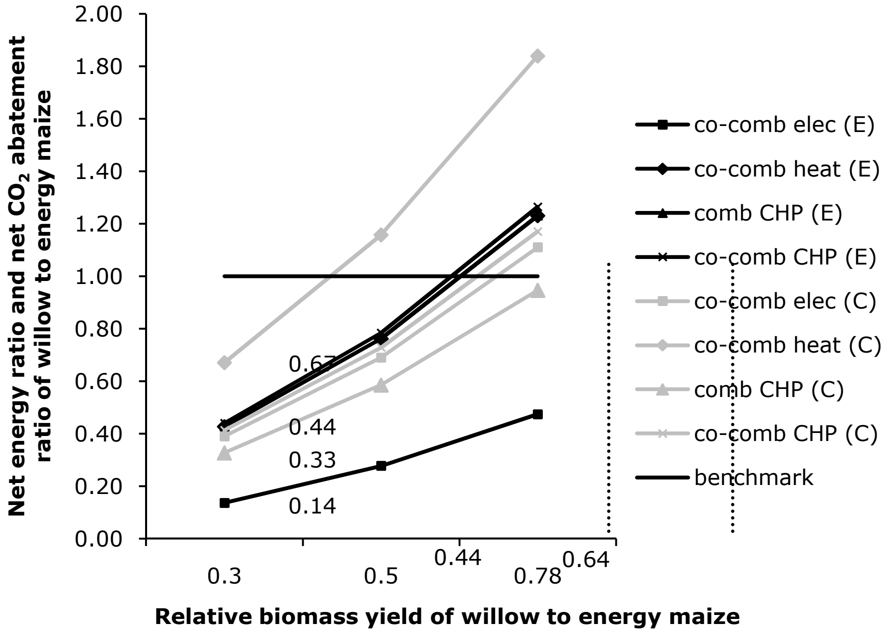


Figure 3-5 Impact of a change in relative biomass yield (dm) of SRC willow to energy maize on the energy ratio (E) and CO_2 abatement ratio (C) for different wood conversion options, *ceteris paribus*

We included the option "co-combustion in a CHP installation", since efficiencies in such installation (η(th)=50%, η(el)=37%) differ from a CHP installation based purely on biomass (η(th)=69%, η(el)=16%). We did not include this option in the main analysis since this is not an option in practice (no existing installation near the case study).

Figure 3-5 demonstrates that SRC of willow could reach the same energy yield per hectare per year as energy maize, given that the relative biomass yield is 0.64, translating into a biomass yield for willow of 13 ton dm ha^{-1} year^{-1}. This lies above the average yield found in literature, but has been reached by one of the non-commercial INBO clones on the experimental field. Moreover, SRC of willow could abate the same amount of CO_2 per hectare per year as energy maize, already when the relative biomass yield is 0.44, translating into a biomass yield for willow of 8.7 ton dm ha^{-1} year^{-1}.

Both net energy potential and CO_2 abatement can be economically valued, with the former a private benefit, while the second an external benefit, and both have an impact on optimal crop choice. Figure 3-4 and Figure 3-5 demonstrate that both benefits do not necessarily go hand in hand: crop-conversion options with an at first sight low net energy production could contribute substantially in our fight against climate change when we consider their potential CO_2 abatement. In August 2009, the United States Environmental Protection Agency published its proposal called Superfund Green Remediation Strategy which outlines strategic recommendations for cleaner site redevelopment (EPA, 2010). The strategy stipulates that green remediation factors might even be included in the evaluation of the economic efficiency. When growing crops for energy production on land, while gradually remediating the soil, a policy suggesting government intervention based on CO_2 abatement might be necessary to improve economic efficiency. This is only allowed because the external benefit of CO_2 abatement is not included correctly in the price of biomass and as such not yet taken into account in economic optimization. As Hughes (2000) already indicated a decade ago, farmers will make rational economic decisions about which crops to plant. The purpose of a correct policy development is then not to intrude in the private costs and benefits so as to force a certain crop in this case, but to internalize factors which are not yet taken into account in the economic analysis. By subsidizing renewable energy based on its external benefit (*i.e.* its CO_2 abatement and valuation) the economics on which the farmer will base its crop decision will be altered the correct way.

Before we can design such policy, the necessary calculations need to be made to see whether government involvement would actually improve economic efficiency and to what extent government should intervene. These calculations were made and discussed in this chapter. How the external benefit of CO_2 abatement from biomass coming from plant-based management of contaminated soils should be economically valued, which economic mechanism is most appropriate, which systems are already put into place and to what extent, and what the implications are of these policies on the phytoremediation technology is studied in Chapter 3.5.

Chapter 3.5 *Economic assessment and policy analysis of CO_2 abatement*

This chapter has been submitted in:
Witters, N., Van Slycken, S., Weyens, N., Thewys, T., Meers, E., Tack, F., Vangronsveld, J. (xxxx) Does phytoremediation redeem to the expectations of being a sustainable remediation technology: a case study II: economic assessment and policy analysis of CO_2 abatement. Biomass Bioenerg.

Abstract

The purpose of this part is to examine the potential economic benefit of energy production of different crops used for plant-based technologies for sustainable land management, and to explore whether existing policies on renewable energy could assist in promoting phytotechnologies as an alternative for conventional energy consuming remediation technologies. Our analysis is based on a case study in the Campine region (Belgium and the Netherlands), where agricultural soils are diffusely contaminated with cadmium, lead, and zinc. Due to the sandy characteristic of the soil and relatively low pH, there is an enhanced risk for uptake of these metals in crops. Regional policy therefore prescribes that food and fodder production should no longer be allowed until soils are remediated. However, the area has such a large extent (700 km²) that conventional remediation is not applicable. Therefore, phytoremediation is suggested as an alternative economically viable, effective, and environmentally sustainable remediation strategy. We first analyzed whether data support this rather loaded, high expectations raising statement. Based on a Life Cycle Analysis (LCA), we examined the energy and CO_2 abatement potential of willow (*Salix* spp), energy maize (*Zea mays*), and rapeseed (*Brassica napus*) after being grown on contaminated land. Further calculations on economic valuation indicated whether subsidizing the use of biomass harvested on contaminated soils would be economically efficient. Our results are based on current Flemish policy and several valuation techniques for CO_2. Our case study indicates energy maize and rapeseed as the economically and energetically most valuable crops. Existing energy subsidies are already reflected in today's crop prices. Therefore, CO_2 abatement potential should not be included again as this would lead to double

counting. Our calculations, based on the “true” price per ton of biomass and the price per GJ to internalize the CO_2 benefit, suggest that current Flemish subsidies for renewable energy production are not sending the right price signals. Implications for the phytoremediation technology are mixed. We found that these true prices are not high enough to encourage renewable energy production, whether contaminated or uncontaminated biomass is used. However, the analysis clearly indicates that including CO_2 benefits would increase the competitive advantage of plant-based technologies over conventional remediation technologies. Our findings support phytoremediation’s label of being a sustainable remediation technology, which could contribute to its introduction on a commercial scale.

3.5.1 Introduction

Phytoremediation is often mentioned as an economically viable, effective and environmentally sustainable remediation strategy (Kumar *et al.,* 1995; Salt *et al*., 1995; Rulkens *et al*., 1998; Susarla *et al*., 2002; Wang *et al*., 2007; Liang *et al.,* 2009). The utilization of the obtained biomass of a phytoextraction cycle as an energy resource is attractive (Chaney *et al.*, 1997) since it can turn phytoextraction into a profit making operation (Robinson *et al.*, 2003; Meers *et al*., 2005a; Vangronsveld *et al.,* 2009; Ruttens *et al.,* 2011) and could moreover contribute to the reduction of global carbon dioxide (CO_2) emissions.

That this potential of phytoremediation crops could become important in the evaluation of technologies, is shown by the recent Superfund Green Remediation Strategy of the US EPA which stipulates that green remediation factors might even be included in the evaluation of the economic efficiency of remediation projects (EPA, 2010). The EU from its side committed to an 8% reduction in CO_2 equivalent emissions, compared to 1990 levels within the context of the Kyoto protocol (UN, 1998). Moreover, the Renewable Energy (RE) Directive promotes an increase in renewable energy to 20% by 2020 and an increase in share of biofuels in transport to 10%. Measures to reach these challenging targets are already implemented in many Member States, such as feed-in tariffs - an obligation on the part of energy suppliers to purchase electricity produced by particular technologies at a specified price guaranteed for a period of time (most member countries) -, the green certificate system (*e.g.* Belgium), tendering (*e.g.* Ireland), and a tax system (additional to other measures, but as the only measure *e.g.* in Malta) (EC, 2005a).

The theory behind these support mechanisms is that (in a perfect market) an individual whose initial desire for a commodity exceeds its price will continue to purchase the commodity until the benefit derived from the last amount purchased equals the price paid for that amount. Externalities are those effects of a production process for a good or service, which are imposed on society or the environment, but are not taken into account in the price of the service or good. These effects may be positive (external benefits) or negative (external costs). Since these external costs and benefits are not included in the price (the mechanism by which producers and consumers are guided), this results in inefficient market allocations. Thus, when externalities are present, this will lead to an over- or underconsumption and over– or underproduction of the good or service, leading to a total social surplus that is diminished with a dead weight loss. As a result, the free market does not produce an efficient level of welfare. Externalities are therefore often referred to as market failures and can be corrected by their incorporation in prices (Merkhofer, 1987; Graves, 2007). When externalities exist, government may be justified in intervening to force a level of welfare that is more socially desirable than the inappropriate one reached through the market. One way for government to change human behavior is through the implementation of mandatory standards and regulations or market oriented incentives, such as subsidies and taxes, designated to force individuals to change their producing and consuming actions (Merkhofer, 1987).

When producing energy (electricity and heat) based on fossil fuels, two categories of external costs arise. A first category refers to costs arising from emissions that cause damage to the environment (*e.g.* acidification) or to people (*e.g.* public health), such as particulate matter, sulphur dioxide (SO_2) and nitrogen oxide (NO_x). Estimated damage costs vary widely across continents. A second category refers to external costs arising from greenhouse gas emissions that lead to climate change. Again, the range of estimates for damage costs is vast. The other way around, renewable energy results in a lot of positive externalities, such as employment, avoided climate change, *etc.* (Saez *et al.*, 1998; Solino *et al.*, 2009).

Life Cycle Analysis (LCA) is an appropriate qualitative methodology to start economic calculations to assess environmental sustainability of energy crops, since it makes environmental effects explicit, which can only facilitate policy making (Hanegraaf *et al.*, 1998). Holmgren (2007) adds to this that studies of energy systems are powerful tools to increase knowledge of the consequences of policy instruments.

Hall and Scrase (1998) state that in industrialized countries the removal of subsidies and tariffs that support unsustainable energy production is a necessary prerequisite for green alternatives to enter the market. The abuse of government support and fiscal measures to not only encourage energetic efficiency, but also the sub-optimal production of electricity has been acknowledged by the European Commission (EC, 2005b).

Page *et al.* (1999) analyzed remediation activities at a contaminated site in Canada using the LCA based approach. Their case study was a parcel of land contaminated predominantly with lead (Pb) arsenic (As), and cadmium (Cd). The remediation approach was excavation combined with disposal. For a remediated site surface of 10,850 m², approximately 2,480 ton CO_2 were emitted, including raw material acquisition (334 ton), site processing (265 ton), and transport (1,880 ton), mainly emitted by diesel fuel use. US EPA (2008) mentions a total CO_2 emission of 271 ton for field machinery and vehicles used for a typical multi-phase extraction project. Table 3-75 provides an overview of average yearly CO_2 emissions for different technologies used in Superfund cleanups (EPA, 2008).

Table 3-75 Overview of estimated total yearly CO_2 emissions (ton) for Superfund cleanups per technology

Technology	*Estimated total yearly CO_2 emissions (ton)*
Pump and treat	323,456
Thermal desorption	57,756
Multi-phase extraction	12,000
Air sparging	6,499
Soil vapor extraction	4,700

Source: EPA (2008)

The potential contribution of the renewable energy potential to the economic assessment of phytoremediation (crops) was acknowledged by Licht and Isebrands (2005), but the authors stayed rather superficial on this subject without making actual calculations. When gradual remediation of soil is combined with energy production, we should acknowledge the potential resulting CO_2 abatement as an external effect of the remediation function. Adding this to the decision function might contribute to the competitiveness of plant-based technologies with conventional remediation technologies. However, this is only allowed when the external benefit of CO_2 abatement is not yet taken into account in biomass prices (*e.g.* through subsidies for renewable energy). As Hughes (2000) already indicated a decade ago, farmers will make rational economic decisions about which crops to plant. The purpose of correct policy development is then not to intrude in the private costs and benefits so as to force a certain crop in this case, but to internalize factors which are not yet taken into account in the economic analysis (Baumol and Oates, 1988). By subsidizing renewable energy based on its external benefit (*i.e.* CO_2 abatement) the economics on which the farmer will base its crop decision will be altered the correct way. In this chapter we present a policy proposal for subsidizing plant-based technologies combined with energy production, based on CO_2 abatement.

Before we can design such policy, the necessary calculations should be made to determine whether phytoremediation is actually capable of delivering the external benefit of CO_2 abatement. Therefore, two large scale experimental fields were installed in Lommel to evaluate the possibilities of cultivation of non-food crops with a main focus on the cultivation of energy crops with high metal accumulating capacities. Crops of interest include short rotation coppice (SRC) of *Salix* spp. (willow), *Populus* spp. (poplar), *Zea mays* (maize), and *Brassica napus* (rapeseed). Calculations on net energy production and CO_2 abatement per hectare per year were made and discussed in Chapter 3.4. This was necessary to analyze whether government involvement could actually improve economic efficiency, and to what extent government should intervene. How the external benefit of CO_2 abatement from biomass used for plant-based management of contaminated soils could be valorized, if these are grounds for subsidizing plant-based technologies, and what the implications are of these policies on phytotechnologies, is studied in this chapter.

3.5.2 Data and methods

3.5.2.1 Economic valuation: government intervention

A control policy is said to be cost-effective if it achieves a given level of aggregate control at minimal total cost, if the burden among n emission sources is divided according to their MAC (marginal abatement cost) function. A control policy is said to be efficient when net benefits of control are maximized. Overall efficiency is then achieved when the marginal cost of control is equal to the marginal damage caused by the pollution for each emitter. Taxes and subsidies are such economic incentive instruments that achieve a given level of environmental quality at least cost, and encourage behavior through price signals rather than through specific target levels of pollution (Hahn and Stavins, 1992). Since the emitter is paying the tax, or the abatement is subsidized, pollution costs are internalized. These kinds of policies are difficult to implement in practice because we need to know the level of pollution at which marginal abatement and marginal damage cost curves cross, and as mentioned, it is hard to determine the marginal damage functions. Therefore, regulatory systems are often preferred as they can limit the emissions of pollutants with relative ease of compliance monitoring and enforcement. They force sources to reach the same goal of pollution control. An example is a maximum emission level e^* of a specified pollutant such as CO_2.

What policy should be preferred for the internalization of CO_2: a tax-, subsidy- or regulatory system? The marginal damage function of CO_2 is a flat function

(Kotchen, M., personal communication, October 2009). It can be graphically shown (Figure 3-6) that under- or overestimating MAC CO_2 when the MDC is flat creates no loss in efficiency in a tax- or subsidy-system, getting the MAC right would have lead to the same height of subsidy or tax. This is however not the case in a regulatory system. “A” represents the loss in total welfare when underestimating MAC (when it is actually MAC^+). The standard should have been set at e^{*+}, but instead, by underestimating MAC, it was set at e^*, leading to an actual abatement up until point e^{st+}. Between e^{st+} and e^{*+}, abatement costs > damage costs, and this results in welfare loss. Likewise, “B” represents the welfare loss when overestimating MAC (when it is actually MAC) when standards are applied. As opposed to a tax-system, the subsidy-system requires money from the regulator, and knowledge of e_u (*i.e.* the initial level of emission) since $s(e_u\text{-}e^*)$ needs to be paid by the regulator. On the other hand, a subsidy system is more easily accepted. A more extended comparison of taxes versus subsidies versus standards can be found in Chapter 1.3.

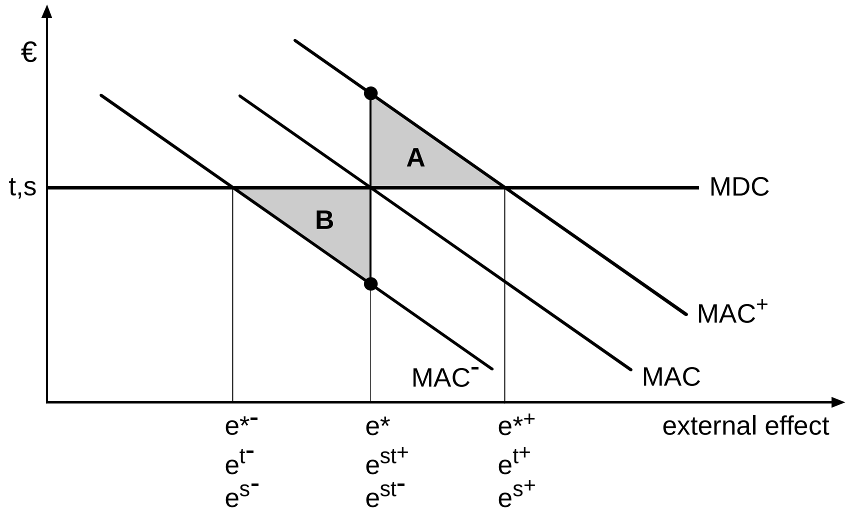


Figure 3-6 Impact of policy choice (standard (st), tax (t) or subsidy (s)) when the marginal abatement cost function (MAC) of CO_2 is an overestimation (when it is actually MAC^-), or underestimation (when it is actually MAC^+), given a flat marginal damage cost function (MDC) of CO_2, with e* the optimal level of external effect for MAC (e*), MAC^+ (e^{*+}), and MAC^- (e^{*-}), and e^t, e^s, e^{st} the actual levels of external effect due to tax, subsidy and standard respectively, for the three MAC curves, resulting in welfare loss A in case of underestimation of MAC combined with a standard, and welfare loss B in case of overestimation of MAC

3.5.2.2 Shedding light on the shadow price of carbon

The market price of carbon is the value of traded carbon emissions rights, imposed by current policy. To arrive at a price of carbon for policy appraisal, the traditional (as for any good) neo-classical approach is estimating the optimal price of carbon. This is the price where the marginal damage cost (MDC) of carbon to climate equals the marginal abatement cost of carbon (MAC) and the point where an economically efficient level of carbon is emitted.

MDC is called the social cost of carbon (SCC) and is a monetary estimate of the cost imposed upon society by the emission of one ton of carbon at some point in time, over the lifetime of that ton in the atmosphere, *i.e.* it is a measure of the carbon externality. Calculating SCC requires quantification of the whole process, linking anthropogenic emissions of greenhouse gases with impacts on social welfare, normalized to impacts on consumption. This is a heroic task performed by integrated assessment models. Because of problems with computing SCC, it is practically impossible to estimate the theoretical optimal price (Dietz, 2007; Ekins, 2007; Price *et al*., 2007), although good estimates can be found amongst others from Chris Hope, William Nordhaus, and Richard Tol (Tol, 2008). Clarkson and Deyes (2002) also give an overview of estimates of SCC in the two previous decades. Using the median of the Fisher-Tippett kernel density for peer-reviewed estimates with a 3% pure rate of time preference and without equity weights, the SCC is \$ 20 ton^{-1} C; *i.e.* € 12.73 at an exchange rate of € 1=\$ 1.57 at the time of publication (Tol, 2008).

In economic appraisal of public investments, the aim is to value changes in the emission of greenhouse gases at their shadow prices (SPC). The shadow price in a given period should equal the change in the policy maker's objective function for a (very small) reduction in carbon emissions in the same period. Whereas the SCC is determined by our understanding and valuation of the damage, SPC is set on the basis of the SCC and is adjusted to reflect the policy and technological environment and targets. SPC can differ from SCC, broadly for two reasons. Firstly, it is calculated for the optimal level of emissions given the objective function and the various constraints. The SCC can be calculated away from the optimum. Secondly, the policy maker's objective function may differ from that assumed in calculating the SCC. That might be the case, for example, because additional factors may influence the policy maker that are not included in the social welfare function on which the SCC is based (Bowen, 2007; Dietz, 2007; Price *et al*., 2007).

An alternative approach for carbon price determination is to find the marginal abatement cost (MAC) of CO_2 that would be required to reduce emissions to reach a global goal of stabilizing carbon concentrations in the atmosphere at a level thought to avoid unacceptably dangerous climate change. Again, we cannot predict with certainty what this level should be, what combination of mitigation techniques and technologies will ultimately be used and where, and how much they will cost when they are used. However, we have far more information today about the costs of these techniques and technologies than we do about the consequences of decades' more warming (SCC), globally.

Estimates all depend on ethical judgments within a particular ethical framework and cannot be determined by purely technical means (Bowen, 2007). Since the beginning of the 90's, researchers are working on a methodology to put actual numbers on external costs, within the ExternE (externalities of energy) project in command of the European Commission. Bickel and Friedrich (2005) found that - for the European Union – damage cost estimates resulted in broad and uncertain ranges.

In accordance with the precautionary principle (since damage costs are highly conservative and only damages with reasonable certainty are included), the European Commission proposes the use of abatement costs in determining the shadow price of carbon, as do other authors (Bickel and Friedrich, 2005; Dietz, 2007; Ekins, 2007; Price *et al*., 2007). The level of abatement is based on Kyoto targets. Costs depend on the chosen policy (the Kyoto protocol defines the target for the EU concerning CO_2 emission reduction, but it does not indicate through which policy the target should be achieved). Policies in Europe traditionally try to balance targets to be reached within countries and a global EU-target, and local measures that need to be taken to reach the local targets and policies on EU-level. For the Kyoto protocol, the EU has developed differentiated targets for each member country. The shadow price of a substance such as CO_2 for a region such as Flanders is equal to the marginal cost to attain the emission goals of that matter for this region (Torfs *et al.*, 2005). Under a full flexibility EU wide allocation of least cost sectoral objectives, studies show a MAC of € 20 ton^{-1}. When each Member State would have to fulfill its objectives on its own, the MAC for Belgium would be € 90 ton^{-1}. Moreover, some countries have taken unilateral actions to reduce CO_2 emissions beyond the target level set by the EU, indicating a higher willingness to pay for a ton CO_2 in these countries. Also, allowing for trading outside the EU would reduce the abatement costs to € 5 ton^{-1}.

Finally, the penalty set in the EU- Emission Trading Scheme is € 40 per ton for the first three years (Bickel and Friedrich, 2005). The EU-ETS allows for cost efficient CO_2 emission reductions for big industrial energy users. Moreover, Member States can use flexible mechanisms like the Joint Implementation or international Emission Trading to meet their emission targets. EUA's (European Union Allowances) are tradable emission credits from the EU Trading Scheme. Each of them carries the right to emit one ton of CO_2. After recovering to € 15 in the last quarter of 2009, prices had slipped back to the € 12-€ 13 range since December 2009, following the disappointing outcomes of the UN's Copenhagen climate conference (www.carbonpositive.net; www.reuters.com).

3.5.2.3 Avoid double counting of external costs of energy production

In producing policy and project appraisals we need to be careful that externalities are not internalized twice. Where policy/project costs already reflect (part of) the social cost of carbon, only the remaining external part should be internalized. Failure to take account of carbon costs that are already internalized will give too much weight to the positive external effect of carbon (Price *et al*., 2007). We made this exercise for Belgium and more specifically Flanders that supports renewable energy through tax exemptions and subsidies, implicitly because of European goals to reduce greenhouse gas emissions.

<u>Taxes in Flanders</u>

According to the European Council Directive 2003/96/EC for the taxation of energy products and electricity, Member States may apply total or partial exemptions or reductions in the level of taxation to (i) electricity generated amongst others from biomass or from products produced from biomass, (ii) energy products and electricity used for combined heat and power generation, (iii) electricity produced from combined heat and power generation, provided that the combined generators are environmentally friendly. In Belgium, the Royal Decision (2006) concerning the use of rapeseed oil as a transport fuel then determines that PPO which is (i) directly sold by a natural person to the end consumer, or is (ii) used for public transport, is 100% exempted from regular taxes on transport fuels[83]. Moreover, every diesel supplier is obliged to also blend at least 4 % (v/v; volume%) of biodiesel (fatty acid methyl ester, FAME) every year (Law of 22.07.2009). When 5 % (v/v) is blended, the biodiesel part is also exempted from taxes.

Certificates in Flanders

In Flanders, green (electricity) certificates and Combined Heat and Power (CHP) certificates have been implemented which, due to their tradability, guarantee that quota are reached by suppliers (minimum quota for green electricity and maximum quota for CO_2). The system of green (electricity) certificates in Flanders consists of two parts. Every electricity supplier is obliged to deliver a specific volume of electricity generated from renewable energy sources and hand in a certain amount of certificates that guarantee that energy is renewable[84] (www.vreg.be). Producers of green electricity receive a certificate for every MWh (net) produced and sell it to electricity suppliers (Energy Decision, 2004). The average price is determined by the market and is on average € 110/certificate (March 2010). The guaranteed minimum price is € 80[85] and the maximum price is equal to the fine of € 125/certificate. This system aims to reach targets as set in Directive 2001/77/EG for Belgium (6% of electricity from renewable resources by 2010) and as confirmed in the RE Directive (2009) (13% of energy from renewable resources in gross final consumption of energy by 2020)[86].

Another official incentive policy involves the support for exploiting a CHP system. This system stimulates that, besides the electricity produced, heat will be recovered, for which government issues CHP certificates. There is no

[83]This doesn't take away the fact that PPO can only be sold on the Belgian market when there is admission from the authorities and when dealers have signed a quality certificate (which guarantees the compliance with at least certain conditions) (K.B. Biofuels, 2005).
[84]Not all renewable energy gets certificates, *e.g.* electricity produced from waste when the Ladder of Lansink is not followed (www.vreg.be).
[85]The New Energy Decree (2009) stipulates a minimum price of € 60/MWh for the technologies studied here.
[86]Those targets from Directive 2001/77/EC that deal with targets and reporting for 2010 should remain in force until the end of 2011 (RE Directive, 2009).

obligation for electricity suppliers to deliver a certain amount of CHP certificates (as opposed to the green certificate system). Conditional on the fulfillment of several qualifications, and depending on the efficiency of the CHP (as compared to a fossil system), the CHP installation receives an average price of € 40/certificate, a guaranteed minimum price of € 27 and a maximum price equal to the fine of € 45/certificate (CHP Decision, 2006; www.vreg.be). Together with the CHP reference Decision (2006), the CHP Decision (2006) lays down the requirements concerning efficiency and support for CHP as explained in Directive 2004/8/EC on the promotion of cogeneration.

Both certificate systems offer economic agents the freedom to decide how to attain objectives, while offering price stimuli (as opposed to regulation), but *e.g.* the green (electricity) certificates will reduce emissions to the level that government decides. This means that rest emissions can be higher or lower than the socially optimal emissions.

The RE Directive (2009) stipulates that public support[87] remains necessary to reach the Community's objectives with regard to the expansion of electricity produced from renewable energy sources, for as long as electricity prices do not reflect the full social (environmental) costs of fossil and benefits of renewable energy sources. In February 2010, the European Commission released a report that concludes that more detailed legislation on sustainability criteria for biomass and waste for energy production is not necessary. However, biofuels used for compliance with European 2020 targets and those that benefit from national support should fulfill sustainability criteria. This includes amongst others a GHG emission saving of at least 35%. For biodiesel from rapeseed, standard

[87] "support scheme" means any instrument, scheme or mechanism applied by a Member State or a group of Member States, that promotes the use of energy from renewable sources by (i) reducing the cost of that energy, (ii) increasing the price at which it can be sold, or (iii) increasing, by means of a renewable energy obligation or otherwise, the volume of such energy purchased. This includes, but is not restricted to, investment aid, tax exemptions or reductions, tax refunds, renewable energy obligation support schemes including those using green certificates, and direct price support schemes including feed-in tariffs and premium payments (RE Directive, art. 2).

GHG emission saving is 38%, for PPO this is 57% and both are thus eligible for public support.

Other support

The CAP (Common Agricultural Policy) Regulation (1782/2003) established common rules for direct support schemes under the CAP, and established certain support schemes. By paying a subsidy on a per hectare basis, instead of on a production basis, it decoupled the link between crop production and subsidies. This means that what farmers grow on their land was no longer only stimulated by the profitability of the crop, but also by the impact of the crop on their land. The CAP Regulation established specific support for energy crops to assist the development of the sector. However, due to (i) recent developments in the bio-energy sector, (ii) the strong demand for biomass for energy on international markets, and (iii) the introduction of binding targets for the share of bio-energy in total fuel by 2020, the Health Check Regulation (73/2009) stipulates that there is no longer reason to grant support for energy crops, and repeals the CAP Regulation.

3.5.3 Results

3.5.3.1 Private economic results

For calculations on the income of farmers from alternative energy crops, we used the adapted gross income (AGI) (Table 3-76).

Table 3-76 Yearly AGI (€ ha^{-1}) and price (€ ton^{-1} fresh matter) of alternative crops for different conversion options

	Energy maize	*Rapeseed-PPO-pers use*	*Rapeseed -PPO-tractor*	*Rapeseed - Biodiesel*	*SRC-electr.*	*SRC-heat*	*SRC-CHP*
AGI’	1,260	1,354	1,195	1,023	111	111	111
price	30	477†	298†	208	28.5	28.5	28.5

AGI=adapted gross income; †3.33 ton seeds are converted into 1,295 liter oil. Value of PPO for personal use is € 1.226 l^{-1}. Value of PPO for tractor is € 0.767 l^{-1}

3.5.3.2 CO_2 abatement

In many European Member States, measures to reach the challenging targets concerning CO_2 emission reduction and minimal levels of renewable energy in the total energy production, to which the EU has committed, have been implemented. All (financial) measures, such as subsidies and tax exemptions, are in theory aimed at supporting Member States in reaching these targets. In the phytoremediation crop choice model we use current biomass market prices which reflect policy measures. We analyze whether these subsidies and tax exemptions to energy conversion installations, which use biomass as a feedstock, correctly reflect the CO_2 abatement potential of biomass. The correct internalization of externalities is what these subsidies and exemptions should be intended for. When this is not the case, this leads to economically inefficient allocations of biomass.

For our case study, green (electricity) certificates, CHP certificates, and tax exemptions are most important, and their correctness for our case study is analyzed, and if necessary corrected. Alternative to certificates and tax exemptions, we would rather suggest an inclusion of the CO_2 abatement potential over the full life cycle of each energy carrier as calculated for each potential crop-technology combination. This abatement is valued by (i) MAC CO_2, and (ii) EUA. In Table 3-77 we calculated for each crop-technology option what the raise in price in biomass in € ton^{-1} (fm and dm), in € GJ^{-1} fossil energy, and in € ha^{-1} should be, if there were no subsidies to promote renewable energy in Belgium/Flanders (*i.e.* if there was no renewable energy support and prices of biomass would not already reflect policy intervention).

Co-combustion of willow to replace for coal-based electricity is the biomass conversion pathway with the most positive external effect (in ton CO_2 GJ^{-1}). However, in Chapter 3.4 we found that co-combustion of willow for heat eliminates a higher amount of CO_2 emissions per ha (ton CO_2 ha^{-1}). This can be explained by the fact that combustion of cokes for heat is energetically more efficient than the combustion of cokes for electricity, high efficiency leads to high energy output per ha. "Price change GJ^{-1}" indicates with what amount in € per GJ the price of the fossil energy carrier should raise to correctly reflect the negative externality of CO_2 emissions. Another interpretation is that it indicates with what amount the price of alternative renewable energy carriers could be reduced to reflect the positive externality of avoiding CO_2 emissions. "Price change ton^{-1}" indicates the allowed price change of biomass per ton because of its potential CO_2 emission abatement, and "price change ha^{-1}" indicates the effect on a per hectare basis (yearly) when CO_2 abatement is valorized. Converting subsidies to stimulate renewable energy results in price changes for willow (on a per ton and per hectare basis) that are higher when willow is used for combustion for heat production, than for electricity production, because of the higher efficiency. Subsidies for energy maize should be highest, and subsidies for rapeseed should be three to four times lower than those for energy maize.

Table 3-78 shows the impact of internalization of CO_2 on original biomass prices (€ ton^{-1} fm) and on original AGI (€ ha^{-1}). If the valuation of CO_2 is based on MAC (€ 20 ton^{-1} CO_2), then for willow this would mean that the current AGI would double to triple. The impact of the internalization on the price of willow (€ ton^{-1} fm) is smaller, but still results in a price that would lie 44-83% higher (than without internalization), depending on the conversion technology for which the wood would be used. The impact on the price per ton is lower than the impact on AGI due to high cultivation costs (especially stool removal), which results in very low AGI.

Table 3-77 Efficient price changes in € (i) per GJ for biomass based electricity, heat and fuel, (ii) per ton fresh matter (fm) and dry matter (dm), and (iii) per ha, based on the net energy production before conversion of the rest product (EO), and based on MAC CO_2 and EUA

			Energy Maize	*SRC(willow)*			*Rapeseed*	
			Digestion	*Electricity*	*Heat*	*CHP*	*PPO*	*Biodiesel*
Biomass	*Yield*	ton fm	60	8.42	8.42	8.42	3.33	3.33
		ton dm	20	4.8	4.8	4.8	3	3
	ton CO_2 GJ^{-1}†	electricity	0.11	0.20		0.11		
		heat	0.06		0.13	0.06		
		fuel					0.08	0.08
MAC CO_2	*price change (€ GJ^{-1})*	electricity	2.30 (average)	3.90 (cokes)		2.30 (average)		
		heat	1.25 (nat. gas)		2.68 (cokes)	1.25 (nat. gas)		
		fuel					1.69	1.69
	price change (€ ton^{-1})	fm	4.51	14.62	23.69	12.59	22.40	26.73
		dm	13.53	25.66	41.56	22.09	24.86	29.67
	price change (€ ha^{-1})		271	123	200	106	75	89
EUA	*price change (€ GJ^{-1})*	electricity	1.38 (average)	2.34 (cokes)		1.38 (average)		
		heat	0.75 (nat. gas)		1.61 (cokes)	0.75 (nat. gas)		
		fuel					1.01	1.01
	price change (€ ton^{-1})	fm	2.71	8.77	14.21	7.55	13.44	16.04
		dm	8.12	15.39	24.94	13.25	14.93	17.82
	price change (€ ha^{-1})		162	74	120	64	45	53

EO = energy output, *i.e.* net energy before preparation, end-use or disposal of the rest product, MAC = marginal abatement cost, EUA = European Union allowance; †Taking into account efficiencies and losses

Table 3-78 Relative impact of internalization of CO_2 based on MAC and EUA (i) as a % of original AGI (€ ha^{-1}), and (ii) as a % of original price (€ ton^{-1} fm)

	Energy maize	*Rapeseed-PPO-pers use*	*Rapeseed - PPO-tractor*	*Rapeseed - Biodiesel*	*SRC-electr.*	*SRC-heat*	*SRC-CHP*
AGI	1,260	1,354	1,195	1,023	111	111	111
Price biomass	30	477	298	208	28.5	28.5	28.5
MAC CO_2							
% AGI	21%	6%	6%	9%	111%	180%	96%
% price	*15%*	5%	8%	13%	51%	83%	44%
EUA							
% AGI	13%	3%	4%	5%	67%	108%	57%
% price	9%	3%	5%	8%	31%	50%	27%

AGI=adapted gross income

Table 3-79 then analyzes the correctness of Belgian subsidies on a per hectare basis. For each crop-conversion option, we recalculated the subsidy (CHP certificates, Green Certificates, and tax exemption) to a per hectare basis. This is then compared with the calculations on the internalization of CO_2 based on MAC CO_2, and EUA. We split the numbers calculated in Table 3-77 in two parts, CO_2 abatement from heat production, and from electricity production, to compare with CHP certificates and Green certificates respectively.

Green (electricity) certificates are awarded to (hybrid) installations for the net amount of electricity produced by biomass (renewable electricity). The net amount of renewable electricity is the sum of electricity used on the site and the electricity put on the net, diminished with electricity used by the installation to prepare biomass for conversion. If other fossil energy sources are used besides electricity, this reduces the amount of net renewable energy. For calculations, we calculated with 90% of the maximum value of a certificate (€ 112.5 MWh^{-1}). Given a yearly electricity production per hectare of 80,590 MJ, this leads to a yearly per hectare support of € 2,519 for digestion of energy maize[88]. For

[88]GCC are only guaranteed for 20 years.

calculations on CHP certificates, we used the 90% value of the administrative fine (€ 40 per certificate). The same hectare of energy maize (producing 68,610 MJ heat per year) receives € 755.50 per hectare support from CHP certificates. The electricity production from willow (31,550 MJ) receives GCC for the total amount of € 986 per hectare. Separate heat production (74,580 MJ) does not receive any subsidies. The combined heat and electricity production from willow gets CHP certificates for 62,150 MJ (€ 388) and green certificates for 12,430 MJ (€ 388).

Diesel used as engine fuel in its pure form is subject to a tax (€ 198.3148 per 1,000 liter), a special tax (€ 153.0063 per 1,000 liter), and a contribution (€ 14.8736 per 1,000 liter). Diesel blended with at least 5 % (v/v) biodiesel is subject to a tax (€ 198.3148 per 1,000 liter), a special tax (€ 134.6966 per 1,000 liter), and a contribution (€ 14.8736 per 1,000 liter). This means a tax exemption of € 366.19 per 1,000 liter pure biodiesel ((153.0063-134.6966)/5%). Rapeseed oil is completely exempted from taxes, special taxes and contributions, resulting in an exemption of € 366.19 per 1,000 l PPO. Given 1,295 l PPO per ha and 1,589 l biodiesel per ha, this leads to a total exemption of respectively € 474 and € 582 for PPO and biodiesel per ha.

Table 3-79 Comparison of current subsidy per crop-conversion option (€ ha^{-1}) with the economic internalization of CO_2 based on MAC CO_2 and EUA (€ ha^{-1} $year^{-1}$, unless otherwise stated)

	Energy Maize	*SRC(willow)*			*Rapeseed*	
Conversion†	*Digestion*	*Electricity*	*Heat*	*CHP*	*PPO*	*Biodiesel*
Subsidy	*CHP+GCC*	*GCC*‡		*CHP+GCC*	*Tax ex*	*Tax ex*
€ ha^{-1}	756 + 2,519	986	0	388+388	474	582
€ ton^{-1} CO_2	177 + 272	160	0	100+272	127	131
MAC CO_2	85 + 186	123	200	77+29	75	89
EUA	51 + 111	74	120	46+17	45	53

†MAC CO_2 (€ 20 ton^{-1}), EUA (€ 12 ton^{-1}); ‡When > 60% biomass is co-combusted in a coal installation (50 MWe), all GCC apply for certificate obligation, otherwise only 50% of certificates count to fulfill obligations

Under the assumption that current subsidies are only intended to abate CO_2 emissions, digestion of energy maize is oversubsidized by at least a factor 12 (Table 3-79). This is mostly attributable to GCC. For the same reason, the electricity and CHP option from SRC of willow are both oversubsidized, but to a lesser extent. Replacement of cokes by heat receives no subsidy, although it should by now be clear that this is one of the options with best perspectives on CO_2 abatement, and thus should deserve stimulation by economic price signals. In Belgium, the option is currently undersubsidized by at least € 120 ha^{-1} year^{-1}. On average, subsidies for biomass based energy production before rest product processing should lie between € 45 and € 271 ha^{-1} year^{-1}.

3.5.4 Discussion and conclusion

In optimizing the economic efficiency of plant-based technologies, we should compare different crops on a per hectare per year basis, not only based on private economic benefits, but also taking into account the external effect of CO_2 abatement per hectare per year of each crop (Chapter 3.4). By integrating this in the private analysis, we are able to make judgments about which energy crop should be grown on contaminated land to lead to highest economic efficiency. This could then be compared with alternative conventional remediation technologies.

If subsidies for renewable energy are already reflected in higher biomass prices, we should not incorporate the external benefit again. According to Schmidhuber (2006), subsidies (and tariffs) are actually reasons why cereals are used in Europe (and the United States) as feedstock for energy purposes, since these tariffs keep prices for feedstock higher than they would otherwise be. A levy in biomass prices due to carbon taxes and subsidies was observed by Ignaciuk *et al.* (2006). In Belgium, the price of the crop does not explicitly depend on the energy conversion technology (and thus the subsidies for these technologies), but prices of agricultural crops have risen in recent years. This should not only be attributed to the production of biomass based energy (*e.g.* also to disappointing harvests, fertilizer prices), but the upsurge in biomass based energy production is definitely one of the contributing factors (Meers *et al.*, 2008b). For our analysis it suffices to say that the demand for biomass for renewable energy production is stimulated by the subsidized system. Therefore, we can state that subsidies are already reflected in current crop prices. Since we do not (yet) know the extent of this, we should not internalize the external benefit of CO_2 abatement in the biomass price again.

Given current biomass prices (including the external benefit of CO_2 abatement), and assuming that effectiveness of remediation is the same for each crop (*i.e.* in the end they reach the same end concentration of contamination), rapeseed and energy maize would be chosen by farmers facing metal enriched land in the context of sustainable land management (and not within the context of actual remediation).

However, three issues should be risen here. A first one is that cost-effectiveness is not enough to compare both technologies, since cost is not the only difference, remediation duration differs as well. And time might be a decisive factor. Main barriers in the development of commercially viable phytoextraction procedures for trace elements remain the long time period required to remediate soil to legal soil standards, and the use/disposal of the contaminated biomass. Based on extraction data from Vangronsveld *et al.* (2009), reducing Cd concentrations in soil from 5 to 2 mg kg^{-1} would take 120 years for willow, and would last 1.5 times longer when growing permanently energy maize, and 3 times longer in case of rapeseed. The implications on crop choice are not straightforward.

There is (i) the trade-off between higher income during remediation and faster remediation, as these do not necessarily go hand in hand (*e.g.* SRC of willow results in faster remediation, but has a lower yearly net revenue than energy maize). The income of an HI activity at an earlier stage might compensate for the lower income during remediation. There is also (ii) the trade-off between different HI activities: they are allowed at different maximum allowed Cd concentrations in soil, and therefore the choice for one or the other HI activity has an influence on remediation duration. This is studied more in detail in Chapter 4.1. Also, in Chapter 4.3, we make the comparison with conventional remediation technologies and calculate, within certain ranges of trace element concentrations, and within the case study conditions of AGI, at what price per ton conventional remediation technologies would be competitive with plant-based technologies.

The second issue is related to the fact that the external effect of CO_2 abatement is not internalized correctly in the biomass price and that this might have consequences, not only on phytoremediation as an alternative technology for conventional technologies, but also on the choice between alternative crops. Our analysis shows that, based on CO_2 abatement, current Flemish subsidies do not completely succeed in sending the right price signals for biomass based renewable energy production[89]. Certificates in Flanders provide financial stimuli for sources in function of environmentally friendly or emission poor fuels but (i) do not guarantee an automatic and full inclusion of external costs of fossil energy or external benefits of green energy and (ii) are not neutral for different technologies concerning the technology's potential positive impact on the environment (Torfs *et al.*, 2005). *E.g.* much objection is raised against subsidies for co-combustion in a coal based electricity installation because these would even render this installation cheaper than a natural gas based electricity installation (Claeys, 2009). The subsidies studied here are much higher than could be justified by CO_2 abatement alone, except for the case of separate heat production. Implications for the phytoremediation technology are not straightforward. The corrections might generate internal shifts amongst alternative crops as one crop might gain advantage over another crop (Table 3-78). Although it might be true that conversion installations are not capable of offering biomass prices completely corrected for CO_2 abatement, this would be true for uncontaminated as well as for contaminated biomass. Moreover, as suggested, including the correct price for CO_2 to fossil prices could make biomass more competitive as an energy fuel input and thus offer opportunities for phytoremediation. The conventional wisdom appears to be that changes in technology are triggered principally by price signals, and as such the internalization of external costs in fossil fuel based energy production might be

[89]Rather they are an attempt to equalize private production costs of biodiesel/PPO, and fossil diesel: the national justification of tax exemptions sent to the European Commission for approval needs an elaborate calculation of the tax exemption based on the difference in private production costs.

necessary to trigger the development of renewable energy technologies (Owen, 2006), and implicitly also the development of phytotechnology as an alternative for conventional remediation technologies. Finally, due to the raise in biomass prices, the plant-based technology would not lose its competitive advantage over conventional technologies although this will have to be studied over a full soil management life cycle.

The third issue is that the external cost we calculated is only a subset of total external costs. According to Hall and Scrase (1998) caution is needed to ensure that other issues are not considered unimportant simply because they are not expressed in monetary terms. We should acknowledge these limitations when coming to policy oriented recommendations based on external cost data (Krewitt, 2002). In literature, other potential externalities of soil remediation relate to biodiversity issues, water control, vegetation filters, biological activity in the soil, carbon sequestration, waste handling, and erosion control (Burger *et al.*, 2004; Licht and Isebrands, 2005; Mirck *et al.*, 2005; Zalesny *et al.*, 2009). The study performed by Van Wezel *et al.* (2007) gives a comprehensive overview of other benefits of less contamination in soil such as positive health effects, improved drinking water, increased property values. Not all of them are technology specific, *i.e.* could contribute to the competitive advantage of phytoremediation over conventional remediation.

It is very hard for new technologies to enter the market. This explains the fact that conventional remediation is still preferred in almost 100% of remediation projects (natural attenuation not taken into account) over green remediation technologies such as phytoremediation (Public Waste Agency Flanders, OVAM, personal communication, January 2011). Given the fact that in a perfect market government intervention would not improve social welfare, subsidies and taxes are only allowed to correct for market imperfections, such as externalities. Costello and Finell (1998) point out that regulatory factors can create technology development opportunities that would not exist in an economic system without any government intervention, *i.e.* government intervention should be allowed if it is intended to stimulate new technologies. However, this raises many issues: from what moment do we consider a new technology market ready, how far does this stimulation go, and when do we stop stimulating? Moreover, internalization of externalities in biomass prices is a nice approach in theory. How this would be translated and monitored practically is not clear yet, since this would result in different biomass prices depending on its end use for energy production, which is logic since the purpose of the subsidies is not to subsidise biomas, but to stimulate the production of renewable energy. This should not withhold us from studying externalities of phytoremediation since it is these findings that support its label as a sustainable technology, which will only stimulate other research and thus the introduction of this technology on a commercial scale.

Section 4

SELECTION OF AN ALTERNATIVE RISK MANAGING CROP

Chapter 4.1 *Crop choice model*

Abstract

Any possible decision is based on the mathematical solution of a function that integrates different criteria. Typically, (environmental) decisions are based on an analysis using a CBA framework, a normative decision model. Due to the construction of a single decision criterion (NPV), it is mathematically possible to find an optimal solution. The outcome of the crop choice model is a combination of one remediation crop and one HI crop for every initial level of contamination (C_0). This is the combination the farmer should grow if he allocates resources efficiently and maximizes his revenue (R).

AGI_{rem} represents yearly income during remediation from time 0 until time x. AGI_{HI} represents yearly income after remediation from time x until ∞. NPV of AGI_{rem} from time x to time 0 is equal to A. NPV of AGI_{HI} from ∞ to time x is equal to B. To sum A and B, the latter is first discounted to time 0. The model then determines x that maximizes the sum of A and B. The outcome depends on trade-offs (i) between higher income during remediation and faster remediation, and (ii) between different HI crops since they are allowed at different maximum allowed Cd concentrations in soil C_q (C_1 versus C_2).

Energy maize is the preferred alternative and safe crop for ranges of Cd concentration in soil between 12 and 10.1 mg kg^{-1}, after which maize for the grain is grown (maximum allowed soil concentration is 10 mg kg^{-1}). Willow is chosen between 3.9 and 3.5 mg kg^{-1} until 3.4 mg kg^{-1}, after which asparagus can be grown, and from 0.6 mg kg^{-1} to remediate until 0.5 mg kg^{-1} from which moment endive is grown. Rapeseed is in competition with energy maize, and our analyses show that energy maize and rapeseed are equally preferred when AGI_{RS} = r_{EM}. Energy maize is the preferred crop for large distances to target (DTT), *i.e.* the difference between C_q and C_0, while willow is preferred for average to small DTT.

Given the contamination range (12-0 mg kg^{-1}), the model only suggests the use of plant-based technologies in 15% of cases. In all other cases, the model

suggests growing the allowed crop. We noticed that a raise in BP_W results in willow to be chosen more often as the remediation crop, and more often actual remediation is suggested by the model. To induce actual remediation (50% of cases), the AGI of willow should at least be € 1,200. Our analyses show that this could not be reached by changing parameters separately, but a simultaneous increase in allowed rotation cycles, an increase in biomass yield, a reduction in costs, and the harvest of leaves might be able to realize this.

These results are coming from the model when it is based on food- and fodder threshold values. However, soil standards exist simultaneously. Given $C_0 > 2$ mg kg^{-1}, which is the BSN above which soil should be remediated according to soil standards, the model never suggests to remediate soil until $C_x < 2$ mg kg^{-1}, simply because this is not the economically optimal solution, given that the model is built on food- and fodder threshold values and that these are valid, equally to soil standards.

Current target values are 1.2 mg kg^{-1} soil. This means that if this condition is fulfilled, every HI crop should be allowed to be grown. Since soil standards are much lower than in the case of food- and threshold values, remediation will at any case take very long. This is a disadvantage for willow, as this crop benefits from short DTT. As a result, energy maize is most often preferred as the alternative crop in case soil standards are applied. Different soil standards only result in a shift in income over time: the exact same income is reached in an earlier stage when soil standards are less strict. Some farmers will benefit from soil standards, while others will benefit from food- and fodder threshold values, depending on their location and thus Cd concentration in the contaminated region.

Using soil standards or food- and fodder threshold values changes the pathway of optimal remediation and crop production and the total welfare reached by applying one or the other standard could be equal, given the correct circumstances related to the distribution of agricultural land in the contaminated region. This could mean that soil standards are determined at a level that is approximately right (by accident). The stricter the soil standard, the more contaminated land should be located in the less contaminated zone to lead to the same economic welfare.

4.1.1 Introduction

Many phytotechnologies for trace elements are at a demonstration level, and relatively few have been applied in practice on large sites. The available data from finished, full-scale projects are still limited. The gap between research and development for the use of phytoremediation at field level is partly due to:
- a lack of awareness by regulators and stakeholders (*e.g.* land owners, tenants);
- a lack of expertise and knowledge by service providers and contractors;
- uncertainties in long-term effectiveness (what is still available?, monitoring); and
- difficulties in the transfer of particular metabolic pathways to productive and widely available plants (Adriaensen *et al.*, 2008).

More data are needed to quantify the underlying economics as a support for public acceptance and to convince policy makers and stakeholders of the use of phytotechnologies as an alternative for conventional remediation technologies (Ruttens and Vangronsveld, 2006; Adriaensen *et al.,* 2008; Vangronsveld *et al.*, 2009). A clear road map for utilisation of phytotechnologies needs to be developed to allow the user to make an informed decision on the most suitable technology for the site requiring remediation or management (Adriaensen *et al.*, 2008). The "sustainable management of trace element contaminated soils" (SUMATECS) program funded by Snowman (2007-2008) showed a clear desire amongst stakeholders for a reliable decision system tool and improved decision support for gentle remediation operations (GRO) and recommended the incorporation of GRO-focused decision support into national guidelines and tools. This decision tool should consider overall life cycle costs, benefits, and risks, specific to site conditions. Also, according to the same program, the remediation of contaminated sites may be feasible with a combined approach: phytotechnologies can become part of sustainable site cleanup together with conventional remediation technologies, or like we interpreted it in Chapter 4.3:

be used in a more acceptable approach with gradual integration of remediation crops in the crop scheme.

In general, a trace element phytoextraction protocol consists of the following elements (Vangronsveld *et al.*, 2009):
(i) cultivation of the appropriate plant/crop species on the contaminated site;
(ii) removal of the harvestable trace element enriched biomass from the site; and
(iii) post-harvest treatment (*i.e.* digestion, pressing, thermal treatments) to reduce volume and/or weight of the biomass for disposal or for its recycling.

Regarding (ii) and (iii), the main barriers to the development of commercially viable phytoextraction procedures for trace elements remain:
- the long time required to remediate soil to standards; and
- the use/disposal of the contaminated biomass.

Regarding (i), a key question which has been in debate since the very beginning of the introduction of the trace element phytoextraction concept is: "Should one use trace element hyperaccumulator plants or high biomass producing plants?" Opposite opinions exist. Chaney *et al.* (1997) favored the former option after they hypothetically calculated Zn removal by hyperaccumulators and high biomass plants and concluded that in any case the use of hyperaccumulators resulted in higher trace element removal. In support of the second option, Kayser *et al.* (2000) reported that the trace element removal capacity of *T. caerulescens* was not very different from that of biomass producing crop species used. Ebbs *et al.* (1997) supported the latter authors after observing ten times higher Cd concentrations in *T. caerulescens*, but also ten times less biomass production as compared to biomass producing crops. Obviously, the choice for the first or second option depends on site characteristics. If crops would suffer from toxicity problems, hyperaccumulators, which in general possess a higher trace element tolerance, should have an obvious advantage. If high biomass crops are chosen, the most suitable one again depends on general site characteristics (Vangronsveld *et al.*, 2009).

To test the economic viability of phytoremediation for the management of a moderately trace element enriched field, we developed an economic decision tool based on the Campine case study. The model does not include hyperaccumulators. We included diverse remediation and traditional crops for reasons explained in detail below. Post-harvest treatment of the biomass only includes energy options (for reasons explained in Section 2). The biomass after conversion needs proper use or disposal.

4.1.2 Data and methods

4.1.2.1 Initial level of soil contamination (C_0)

As a maximum value for the initial level of soil contamination (C_0) we use 12 mg kg^{-1} since this is a common highest measured value in the heavily contaminated zone surrounding the smelters.

4.1.2.2 Final level of soil contamination

As mentioned in Section 2, two policies concerning soil remediation apply simultaneously. The first policy is based on soil standards (Table 1-5). In the base case, the business model for phytoremediation builds on the second policy, food- and fodder threshold values (Table 1-6, and Table 1-7).

Within the BeNeKempen project, the cultivation advice is an attempt to indicate at what Cd concentration in soil these maximum concentrations in different HI crops are respected, given different pH levels in soil. Simultaneous sampling of soil and crops was planned within the BeNeKempen project. All paired data consisted of a simultaneous measurement of Cd in soil and crop, together with a level of pH. Performing a regression analysis on all paired data, the BeNeKempen project defined for different Cd concentrations in soil and for different pH levels, the probability of exceeding legal food- and fodder threshold values for 16 agricultural crops, maize, and grass. These data are transformed into Table 4-1 which represents for each of the HI crops and maize the maximum Cd concentration in soil for which the crop can be safely grown, *i.e.* without exceeding Cd threshold values for food and fodder use in 50%, 10%

(base case), and 5% of measurements. *E.g.* the food threshold value for maximum allowed Cd concentration in endive is 0.2 mg kg^{-1} fm (Regulation n° 1881/2006), or 3.5 mg kg^{-1} dm (Smolders *et al*., 2007). At a soil pH of 5, this translates into a maximum Cd concentration in soil of 0.5 mg kg^{-1} soil so that no more than 10% of harvested endive exceed food- and fodder threshold values. For maize the maximum allowed Cd concentration is 1 mg kg^{-1} (12% water) (Fodder Decision), 1.14 mg kg^{-1} dm (Smolders *et al*., 2007), and 0.27 mg kg^{-1} fm (76% water), whether for silage or for grain. At a soil pH of 5, this however translates into very different maximum Cd concentration in soil of 1.15 mg kg^{-1} soil for silage maize and 10 mg kg^{-1} soil for grains, so that no more than 10% of harvested maize for silage and maize for grain respectively would exceed food- and fodder threshold values (Vangronsveld, J., personal communication, February 2010).

Table 4-1 Estimated maximum concentrations of Cd in soil (mg kg^{-1}) so that (i) the average, (ii) no more than 10% and (iii) no more than 5% of crops exceed food and fodder threshold values after harvest[†]

	Average				*<10%*				*<5%*			
pH_{KCl}	*4.5*	*5*	*5.5*	*6.5*	*4.5*	*5*	*5.5*	*6.5*	*4.5*	*5*	*5.5*	*6.5*
Potato	19	26	34	59	1.5	2	2.6	4.3	0.8	1	1.3	2.2
Endive	1.0	1.8	3.5	12.4	0.3	0.5	0.9	3.2	0.2	0.3	0.6	2.3
Celeriac	0.2	0.3	0.4	1.1	0.2	0.2	0.3	0.4	0.2	0.2	0.2	0.3
Cucumber	27.4	27.4	27.4	27.4	9.5	9.5	9.5	9.5	7.2	7.2	7.2	7.2
Cabbage	3.9	3.9	3.9	3.9	1.7	1.7	1.7	1.7	1.7	1.7	1.7	1.7
Oignon	4.7	4.7	4.7	4.7	1.8	1.8	1.8	1.8	1.8	1.8	1.8	1.8
Peas	1.6	1.6	1.6	1.6	0.8	0.8	0.8	0.8	0.8	0.8	0.8	0.8
Asparagus	5.9	5.9	5.9	5.9	3.4	3.4	3.4	3.4	3.4	3.4	3.4	3.4
Beans	4.4	6.3	9.0	18.1	1.9	2.7	3.8	7.4	1.5	2.1	3	5.8
Scorzonera	<0.1[‡]	0.1	0.3	2.7	<0.1	<0.1	0.1	0.7	<0.1	<0.1	<0.1	0.5
Lettuce	1.9	2.4	3.1	5.1	0.5	0.6	0.8	1.3	0.3	0.4	0.5	0.9
Spinach	1.2	1.3	1.5	1.8	0.3	0.4	0.4	0.5	0.2	0.3	0.3	0.4
Tomato	15.8	19.3	23.5	34.9	1.6	2.1	2.6	3.9	0.9	1.1	1.4	2.2
Carrots	1.0	1.3	1.7	3.0	0.3	0.3	0.5	0.9	0.2	0.2	0.3	0.5
Celery	0.3	0.4	0.5	1.0	<0.1	0.1	0.2	0.3	<0.1	<0.1	0.1	0.2
Leek	0.7	1.2	2	5.6	0.2	0.3	0.5	1.3	0.1	0.2	0.3	0.9
Grass	>10[§]	>10	>10	>10	>10	>10	>10	>10	>10	>10	>10	>10
Maize (corn cob)	>10	>10	>10	>10	>10	>10	>10	>10	>10	>10	>10	>10
Maize (total)	4.9	5.9	7.2	10.5	0.9	1.15	1.4	1.9	0.6	0.7	0.9	1.3

Source: Smolders *et al.* (2007); [†]Based on Geysen, D., (personal communication, 2007) and Ruttens *et al.* (2008), we use a pH value of 5; [‡]For calculations we use maximum concentrations of Cd as such: in this example, we use 0.1 as if this would be the maximum concentration; [§]We use 10

4.1.2.3 Remediation duration

$REM_i = A \cdot d \cdot \rho \cdot (C_q - C_0) = Q_q - Q_0$ (Eq. 19)

$REM_i = A \cdot t_i \cdot BP_i \cdot E_i$ (Eq. 20)

$t_i = (Q_q - Q_0)/(BP_i \cdot E_i)$ (Eq. 21)

The total amount of metals to be removed per hectare by crop i (REM_i) is the difference between the amount of metals initially present in the soil of the polluted site (Q_0) and the amount of metals in soil for which it is allowed to grow HI crops (Q_q), or in general, the final amount of metals in soil that is economically most viable (Q_x). The total initial amount of metals (Q_0) is calculated as the product of soil depth (d), soil density (ρ), the area surface (A), and initial metal concentration in soil (C_0). Likewise, the final amount of metals in soil (Q_q) is based on C_q (Eq. 19).

REM_i is also equal to plant biomass production per hectare per year (BP_i) multiplied with the metal content in the harvested plant biomass (E_i), the number of years of plant growth (t_i), and the area surface (A) (Eq. 20). For each crop, BP_i is the sum of the different plant parts, and E_i is the average concentration level in the total plant based on the metal concentration in each of the plant parts, and the m% of each of the plant parts. Substituting REM_i from (Eq. 19) in (Eq. 20), the time (t_i) needed to remediate one hectare of soil up to a level C_q can be calculated (Eq. 21) (Japenga *et al.*, 2007).

Calculations for obliged Cd removal (g ha^{-1}) (REM_i) in (Eq. 19) are based on a surface of 1 ha (A), 40 cm soil depth[90] (d), a soil density of 1.6 ton m^{-3} (ρ), initial Cd concentrations in soil (mg kg^{-1} soil) (C_0), and end concentrations of Cd in soil (mg kg^{-1} soil) to comply with food standards (C_q). Extraction capacities for remediating crops (E_i) originate from field trials in the Campine region (Table

[90]Contamination is homogeneously spread within a 40 cm layer due to ploughing. Although roots of maize and rapeseed do not reach this deep, contamination is homogenized every year over this 40 cm layer due to ploughing, resulting in maize and rapeseed cleaning up the 40 cm top layer. For SRC of willow, roots reach a depth of 50 cm but contamination is only detectable in the upper 40 cm.

3-1, Table 3-2, and Table 3-3). A correct calculation of remediation duration is primordial for the final outcome as a shorter remediation duration will allow growing HI crops earlier in time.

The remediation duration (t_i) of crop i in years is presented in (Eq. 21) as the amount of obliged Cd (g ha^{-1}) removal ($Q_q - Q_0$) divided by the product of yearly biomass potential of crop i (ton dm ha^{-1} $year^{-1}$) (BP_i) and concentration of Cd in crop i (mg kg^{-1} dm) (E_i). BP_i and E_i are here considered as constant. However, both depend on soil characteristics and available metal concentrations in soil and might change as metal concentration in soil decreases. Therefore, in practice, this basic linear calculation of t_i which assumes that BP_i and E_i remain constant over time, might be an underestimation of reality, *i.e.* actual remediation might take longer.

Total metal removal rate and resulting remediation duration depend on soil characteristics, metal contamination, available metals, crop metal extraction, and crop biomass production (Koopmans *et al.*, 2008). When metals are removed, the total metal contamination reduces, as well as the concentration of available metals. This relation can be linear but is in most cases logarithmic, meaning that the concentration of available metals reduces faster than the level of total metals in soil, reaching a limit amount of available metals. Over time, metal concentration in the plant (E_i) is then affected by the amount of available metals in soil. However, some authors describe a replenishment of the available metal pool (Van Nevel *et al.*, 2007). This is also the case for sandy soils (Vangronsveld, J., personal communication, March 2010). Also, biomass production of the plant (BP_i) might change over time. In our case, the biomass potential of willow, BP_W, might increase after several years within a rotation cycle, whereas the biomass potential of energy maize and rapeseed, BP_{EM} and BP_{RS} might decrease in time (due to nutrient depletion). Moreover, the depth of the rooting zone might be a factor that influences Cd concentration in plants, as concentration found in plants might differ according to root depth. Finally, there could also be an output of metals, like Cd and Zn leaching losses to the groundwater (van der Grift and Griffioen, 2008).

Although a discussion on the non-linearity assumption can be found in 4.1.4.4, we decided to base the model on linear metal extraction by crops because (i) the Campine soil is a sandy soil, so chemical equilibriums restore fast when available metal pools are exhausted, and (ii) we have no long-term data available on the relation between extraction of metals and soil concentrations. Long-term field experiments are not available (yet) for the Campine region and estimates of the remediation duration have to be based on model calculations. Our estimates of the remediation duration (Eq. 3) to realize the objective end point of Cd in soil are based on constant rates of metal accumulation into plant roots (E_i), based on short-term experiments. This is according to widely available literature.

4.1.2.4 Overview of economics of alternative crops

Table 4-2 AGI, AGI', and r per hectare of alternative crops for different conversion options

	Energy maize	*Rapeseed-PPO-pers use*	*Rapeseed-PPO-tractor*	*Rapeseed-Biodiesel*	*SRC-electr.*	*SRC-heat*	*SRC-CHP*
AGI	1,260.00	1,542.97	1,064.83	548.64	110.72	110.72	110.72
AGI'	1,260.00	1,354.32	1,194.94	1,022.88	110.72	110.72	110.72
EC_M	(217.32)	0	0	0	(0.46)	(0.46)	(0.46)
$r^{\dagger}$	1,042.68	1,209.44	1,050.06	878.00	110.26	110.26	110.26

AGI = adapted gross income, AGI' takes into account crop rotations, r = AGI' adapted for external cost of metals; $^{\dagger}r_{RS}$ changes compared to AGI'_{RS} since rapeseed is grown in rotation with energy maize, and thus also has the effect of the metals in energy maize

In Chapter 3.4 we examined the energy and CO_2 abatement potential of willow (*Salix* spp), energy maize (*Zea mays*), and rapeseed (*Brassica napus*) after being grown on metal enriched land based on a Life Cycle Analysis (LCA). All crops offer the potential to reduce CO_2 emissions (Table 4-3), resulting in net benefits of CO_2 abatement. Further calculations on economic valuation indicated whether subsidizing the use of biomass harvested on contaminated soils would be economically efficient. Our results are based on current Flemish policy and several valuation techniques for CO_2. Existing energy subsidies are already reflected in today's crop prices. Therefore, CO_2 abatement potential should not be included again as this would be double counting. This means that the (external cost and) external benefit of CO_2 abatement (EC_{EN} and EB_{EN} respectively) should not be added again to the model for any of the crops.

Table 4-3 Comparing Belgian subsidy and MAC CO_2 for internalizing CO_2 abatement potential (based on EO) remediating crops

	EM	*SRC*			*RS*	
	Digestion	*Electricity*	*Heat*	*CHP*	*PPO*	*Biodiesel*
Avoided CO_2 (kg)	12,536	5,317	9,135	4,460	2,584	2,936
MAC CO_2 (€)	85 + 185	123	200	77 + 29	75	89
Subsidy (€)	756+2,519	986	0	388+388	474	582

EM = energy maize, SRC= SRC of willow, RS = rapeseed

Moreover, in Chapter 3.3, comparing the economically most viable option for uncontaminated biomass with the economically most viable option for contaminated biomass, we found that the presence of metals has an effect on the use of energy maize (and a negligible effect for willow). We based our economic calculations on existing legislation. Since we calculated AGI with the general price for energy maize, and we also took the price for uncontaminated wood to calculate the AGI of willow, we are sure that the external cost of metals is not yet taken into account in the phytoremediation business model. Therefore, we calculate r, this is the AGI' including the external costs (benefits) of metals (EC_M and EB_M) (Table 4-2).

4.1.3 Model

4.1.3.1 General overview

Any possible decision is based on the mathematical solution of a function that integrates different criteria. Typically, (environmental) decisions are based on an analysis using a CBA framework, a normative decision model. Due to the construction of a single decision criterion (NPV), it is mathematically possible to find an optimal solution. The outcome of the phytoremediation model is a combination of one remediation crop and one HI crop for every initial level of contamination (C_0). This is the combination the farmer should grow if he allocates resources efficiently and maximizes NPV(AGI).

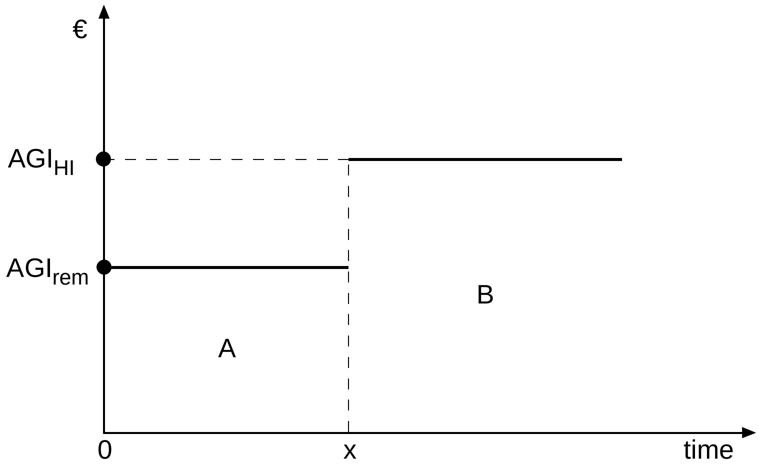


Figure 4-1 Trade-off between fast and slow remediation crop with the income during (A) and after (B) remediation, based on the Adapted Gross Income (AGI) of remediation crops (rem) and high income activities (HI)

In Figure 4-1, AGI_{rem} represents yearly income during remediation from time 0 until time x. AGI_{HI} represents yearly income after remediation from time x until ∞. NPV of AGI_{rem} from time x to time 0 is equal to A. NPV of AGI_{HI} from ∞ to time x is equal to B. To sum A and B, the latter is first discounted to time 0. The model then determines x that maximizes the sum of A and B. This includes the choice of remediation crop, the choice of HI activity, and the end concentration of contamination (mind that this does not necessarily have to be equal to zero) which will lead to an optimal remediation duration x. The outcome depends on the trade-offs discussed in Figure 4-2.

Two trade-offs exist. There is (i) the trade-off between higher income during remediation and faster remediation as these do not necessarily go hand in hand. Given a level of Cd concentration in soil equal to C_q from which a given HI crop q is allowed to be grown, choosing for remediation crop "a" will result in faster remediation than choosing for remediation crop "b" ($t_1<t_2$). However, there is no reason to believe that choosing for "a" will also result in a higher total revenue than choosing for "b" since the faster introduction of the high income crop after remediation might not be able to compensate for the lower income (of "a"

compared to "b") during remediation. *E.g.* SRC of willow results in faster remediation, but has a lower AGI than energy maize).

There is also (ii) the trade-off between different HI crops since they are allowed at different maximum allowed Cd concentrations in soil C_q (C_1 versus C_2). Given a remediation crop "a", the choice for the HI crop has an influence on remediation duration. Again, the income of a HI crop grown later (starting from time t_2 at concentration C_2) has to compensate for the lower income during remediation and there is no necessary positive relation between lower C_q and higher income of the HI crop. It is possible that $AGI_{rem} = AGI_{HI}$, *e.g.* when we start from $C_0 = C_q$ at which point it is already possible to grow a HI crop q. It might even be economically more efficient at C_0 to grow a HI crop with $C_q > C_0$. Our model searches for the optimal C_x.

(i) (ii)

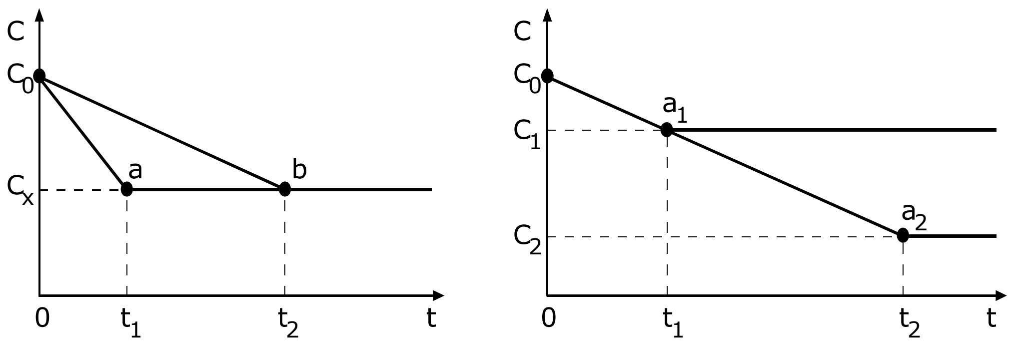


Figure 4-2 Potential trade-offs in phytoremediation model, with (i) trade-off between fast remediation and high income during remediation, and with (ii) trade-off between high C_x and high income for HI crop

The following formulae are used.

The present value of a one-time (at time x) cash flow a:

$$NPV(a) = a/(1 + i)^x \qquad \text{(Eq. 22)}$$

The present value of a series of x terminable annual cash flows a:

$NPV(a) = [(1 + i)^x - 1]/[i \cdot (1 + i)^x] \cdot a$ (Eq. 23)

$NPV(a) = [(1 - 1/(1 + i)^x)/i] \cdot a$ (Eq. 24)

The present value of a permanent annual cash flow a:

$NPV(a) = a/i$ (Eq. 25)

The present value of a permanent annual cash flow a, starting from time x[91]:

$NPV(a) = (a/i)/(1+i)^x$ (Eq. 26)

4.1.3.2 Yearly accumulation per crop per hectare: $BP_i \cdot E_i$

E_i is the average extraction capacity of remediation crop i (mg kg^{-1} dm), based on the mass percentage (m%) of each of the organs in total crop biomass production and metal concentrations in each of the organs, and BP_i is the total biomass yield (ton dm ha^{-1} $year^{-1}$). Maize and rapeseed are harvested in total, willow can be harvested with (g=1), or without (g=0) leaves. Data on biomass production and extraction capacities are detailed for different crop parts (Table 3-1, Table 3-2, and Table 3-3).

Energy maize $0.34 \cdot 0.09 \cdot BP_{EM} + 0.24 \cdot 0.4 \cdot BP_{EM} + 0.68 \cdot 0.06 \cdot BP_{EM} + 1.26 \cdot 0.26 \cdot BP_{EM} + 3.20 \cdot 0.19 \cdot BP_{EM}$

Willow $g \cdot 40 \cdot 0.2 \cdot BP_W + 25 \cdot 0.8 \cdot BP_W$

Rapeseed $0.81 \cdot 0.57 \cdot BP_{RS} + 5.27 \cdot 0.43 \cdot BP_{RS}$

BP_{EM}=20, BP_W=6 and BP_{RS}=5.2 ton dm ha^{-1} $year^{-1}$. This results in accumulation levels of 22, 170 and 14 g ha^{-1} $year^{-1}$ for energy maize, willow (g=1) and rapeseed respectively. When willow leaves are not harvested (g=0), accumulation is reduced with 29% to 120 g ha^{-1} $year^{-1}$.

[91]This assumption holds when we assume that x is small compared to ∞, and thus ∞ - x = ∞

4.1.3.3 *Total accumulation by crop i: $REM_i = Q_0 - Q_q = A \cdot d \cdot \rho \cdot (C_0 - C_q) = A \cdot t_i \cdot BP_i \cdot E_i$ (Eq. 19) and (Eq. 20)*

Total accumulation ($Q_0 - Q_q$) is defined as the total amount of Cd to be removed to go from a Cd concentration in soil of C_0 to C_q. C_q is an end concentration of Cd in soil for a crop HI_q to comply with food threshold values (or in the sensitivity analysis to comply with soil standards). C_q for each HI crop (HI_q) is given in Table 4-1. C_q based on soil standards are given in Table 1-5. Calculations on the total amount of metals to be removed by crop i are based on a per hectare basis (A = 1 ha), a 40 cm soil depth(d), a soil density of 1.6 ton m^{-3} (ρ), and assuming linear extrapolation. This is equal to the total amount that could be removed over time t by crop i, with a biomass potential (BP_i) and with a metal extraction potential (E_i).

Energy Maize $6{,}400 \cdot (C_q - C_0) = (0.34 \cdot 0.09 \cdot BP_{EM} + 0.24 \cdot 0.4 \cdot BP_{EM} + 0.68 \cdot 0.06 \cdot BP_{EM} + 1.26 \cdot 0.26 \cdot BP_{EM} + 3.20 \cdot 0.19 \cdot BP_{EM}) \cdot t_{EM}$

Willow $6{,}400 \cdot (C_q - C_0) = (g \cdot 40 \cdot 0.2 \cdot BP_W + 25 \cdot 0.8 \cdot BP_W) \cdot t_W$

Rapeseed $6{,}400 \cdot (C_q - C_0) = (0.81 \cdot 0.57 \cdot BP_{RS} + 5.27 \cdot 0.43 \cdot BP_{RS}) \cdot t_{RS}$

From these equations we are able to calculate the time (t_i) it would take for crop i to reduce Cd concentrations in soil from C_0 to C_q.

4.1.3.4 *Remediation duration: $t_i = (Q_0 - Q_q)/(BP_i \cdot E_i)$*

Table 4-4 represents for each remediating crop the total remediation duration if that crop were to be used to go from the initial Cd concentration (C_0) to the end Cd concentration (C_q) in soil.

Table 4-4 Total remediation duration (t_i) for energy maize, SRC of willow, and rapeseed to remediate soil with initial concentration C_0 to end Cd concentration C_q

Energy maize	*Willow*	*Rapeseed*
If (	*If (*	*If (*
C_0>C_q ;	*C_0>C_x ;*	*C_0>C_q ;*
[(C_0-C_q) · 6,400/22];	*((C_0-C_q) · 6,400)/12;*	*[(C_0-C_q) · 6,400/14];*
0)	*0)*	*0)*

Adjustments are made as follows:

Rapeseed + energy maize

Rapeseed is grown in rotation with energy maize (1 year rapeseed, followed by 2 years energy maize). Therefore, the average REM_{RS} of rapeseed is 19 g ha^{-1} $year^{-1}$.

Willow + Energy maize

If we assume that SRC of willow is grown over full life cycles (z being the number of cycles) of 22 years (1 year of initiation + 21 years growth)[92] and if the number of remediation years is not a multiple of 22, the model could act as follows:

(i) the rest of the years to reach C_q are calculated back to a soil concentration of Cd, from which energy maize is used to reach C_q;

(ii) one could argue for SRC of willow is grown during another full life cycle (total cycles = z+1), to reach a concentration that lies below C_q. This second option is present in the model, as for each C_0, there is an end concentration that needs z+1 cycles instead of z (this is just another C_o-C_q combination in the model). *E.g.* to go from C_0 = 11.6 to C_q = 11.4 would take 7 years with willow. Therefore, instead of growing willow for 7 years, we grow willow for 22 years, until C_q = 11.

[92]The model assumes that the year of initiation also remediates soil.

Table 4-5 Impact of rotation schemes on remediation duration (years) for different (C_0-C_q)

Alternative crop	*(C_0-C_q) = 3*	*(C_0-C_q) = 2*	*(C_0-C_q) = 1*
EM	880	587	293
RS	1,369	913	456
RS + EM	999	666	333
W (g=0)	160	107	53
W (g=1)	114	76	38
W + EM (g=0)	22*7 + 33 = 187	22*4 +103 = 191	22*2 + 51 = 95
W + EM (g=1)	22*5 + 33 = 143	22*3 + 79 = 145	22*1 + 124 = 146
W + W (g=0)	22*7 + 6 = 160	22*4 + 19 = 107	22*2 + 9 = 53
W + W (g=1)	22*5 + 4 = 114	22*3 + 10 = 76	22*1 + 16 = 38

EM = energy maize, RS = rapeseed, RS + EM = 1 year rapeseed followed by 3 years energy maize, W = willow, W + EM = willow followed by energy maize, W + W = willow followed by willow (not a full cycle)

Willow + willow

As can be seen in Table 4-5, the previous assumption of full willow life cycles has large consequences for the remediation duration for willow. It is based on economic intuition since it seems economic nonsense to grow willow for *e.g.* 6 years, given the high planting and stool removal costs. However, this does not take into account the fact that growing energy maize to remove the remaining contamination results in much higher total remediation durations compare "W + EM" with "W + W". Consequently, HI crops can be grown much later. This might have an economic effect for short remediation durations, *i.e.* where the income of the HI crop actually could have been felt. We could therefore argue that the assumption of full life cycles of willow comes too soon, and that the decision should be based on an economic analysis. These adjustments result in Table 4-6. AGI when not growing willow in a full rotation cycle of 22 years (AGI*) is shown in Figure 4-3. In the model we therefore use "willow + willow" instead of "willow + energy maize".

Table 4-6 ADJUSTED total remediation duration (t_i) for energy maize, SRC of willow, and rapeseed to remediate soil with initial concentration C_0 to end Cd concentration C_q †

Willow + energy maize	*Willow + willow*	*Rapeseed + energy maize*
If (	*If (*	*If (*
$C_0>C_q$;	$C_0>C_q$;	$C_0>C_q$;
$[z \cdot 22 + ((C_0-C_q) \cdot 6{,}400 - (22 \cdot z \cdot 120)) /(22)];$	$[z \cdot 22 + ((C_0-C_q) \cdot 6{,}400/12 - (z \cdot 22))];$	$[(C_0-C_q) \cdot 6{,}400/19];$
0)	*0)*	*0)*

†z is the number of full SRC cycles (22 years) and is the integer of $[((C_0-C_q) \cdot 6{,}400/120)/22]$

Moreover, the model does not take into account metal accumulation by HI crops. Also, the model assumes that the enrichment with metals by reapplying the digestate from energy maize is negligible (the amount of reapplied digestate is very low).

4.1.3.5 *AGI' with corrections for externalities based on current legislation*

For each alternative crop, AGI' is corrected for external costs and benefits of renewable energy production and metal waste disposal (through current policy/legislation)[93], resulting in r_{EM}, r_W and r_{RS}.

$$r_{EM} = AGI'_{EM} - EC_{M\ EM} - EC_{EN\ EM} + EB_{M\ EM} + EB_{EN\ EM}$$
$$r_W = AGI'_W - EC_{M\ W} - EC_{EN\ W} + EB_{M\ W} + EB_{EN\ W}$$
$$r_{RS} = AGI'_{RS} - EC_{M\ RS} - EC_{EN\ RS} + EB_{M\ RS} + EB_{EN\ RS}$$
$$r_{HI} = AGI'_{HI}$$

With EC_M and EB_M = external cost (benefit) of metal waste disposal
EC_{EN} and EB_{EN} = external cost (benefit) of CO_2 abatement

[93]In the externality assessment we do not take into account energy use for growing HI crops.

4.1.3.6 NPV(r) = R

r_{rem} is discounted at a social discount rate of 4% over time t, with t= f(remediation crop, C_0, C_q), using the formula in (Eq. 25) resulting in R_{rem} (Table 4-7, Table 4-8, Table 4-9, and Table 4-10). AGI'_{HI} (=r_{HI}) is discounted at a social discount rate of 4%, over time ∞ - t, resulting in R_{HI}.

Given an initial level of contamination, the longest remediation duration (t_{max} = max t_i) could have been chosen as the discounting period. Remediation crops (i) resulting in shorter remediation periods (t_i) would then offer the advantage of growing HI crops for the rest of the discounting period (t_{max}-t_i). However, as the longest remediation period depends on the outcome of the model, we opt for a general discounting period, *i.e.* infinity (∞).

$R_{EM} = NPV(r_{EM}) = NPV(AGI'_{EM} - EC_{M\ EM} - EC_{EN\ EM} + EB_{M\ EM} + EB_{EN\ EM})$
$R_W = NPV(r_W) = NPV(AGI'_W - EC_{M\ W} - EC_{EN\ W} + EB_{M\ W} + EB_{EN\ W})$
$R_{RS} = NPV(r_{RS}) = NPV(AGI'_{RS} - EC_{M\ RS} - EC_{EN\ RS} + EB_{M\ RS} + EB_{EN\ RS})$
$R_{HI} = NPV(r_{HI}) = NPV(AGI'_{HI})$

The NPV for remediation crops is based on (Eq. 25), the NPV for HI crops on (Eq. 26).

4.1.3.7 Max R_{rem} + R_{HI}

The next step in the model is to maximize the sum of R_{rem} and R_{HI}. The maximizing step consists of two phases. First, for every C_0-C_q combination, the model decides whether energy maize, rapeseed, or willow will lead to the highest R.

Max (R_{EM}, R_W, R_{RS})

Second, the model calculates for every C_0 the economically most viable C_q (=C_x), *i.e.* the one that would lead to the highest R (accompanied by the remediating crop from the first step).

If [

max=EM ;

if [max=W ; if (max= RS; EM or RS or W; EM or W) ; if (max= RS; EM or RS; EM)] ;

if [max = W ; if (max= RS; W or RS; W) ; RS]

]

t_i represented remediation duration for remediation crop i, while x is now the calculated economically optimal last year of remediation, and C_x is the economically optimal C_q. HI crops are grown from time x+1 until infinity, *i.e.* over a time period ∞-x. To add to R_{rem}, R_{HI} is discounted from year x to year 0 (Table 4-7, Table 4-8, and Table 4-10). To arrive at an average yearly income per hectare (r in € ha^{-1} y^{-1}), this sum is then multiplied with i (Eq. 25).

Table 4-7 R with energy maize as alternative crop

	C_x
If (	
$C_0<C_x$;	
$R_{previousC0}$;	
	(
	$[([1-(1/(1+i)^x)]/i) \cdot AGI_{EM}] + [(AGI_{HI}/i)/(1+i)^x]$
	)
)	

Table 4-8 R with willow followed by energy maize as alternative crop†

	C_x
If (	
$C_0 < C_x$;	
$R_{previousC0}$;	
	(
	$[([1-(1/(1+i)^{22z})]/i) \cdot AGI_W] + ([([1-(1/(1+i)^{p'})]/i) \cdot AGI_{EM}]/(1+i)^{22z}) +$
	$[(AGI_{HI}/i)/(1+i)^{x'}]$
	)
)	

†x=22z+p; with z is integer and p is rest. p' is the extra time it would take when after full cycles of willow, soil is polished with energy maize, with x' = z · 22+p'. Although energy maize is grown on a year per year basis, we use the p' that exactly leads to the C_x, so p' might not be an integer

Table 4-9 R with willow followed by an incomplete willow rotation cycle†

	C_x
If (	
$C_0 < C_x$;	
$R_{previousC0}$;	
	(
	$[([1-(1/(1+i)^{22z})]/i) \cdot AGI_W] + ([([1-(1/(1+i)^{p})]/i) \cdot AGI^*_W]/(1+i)^{22z}) +$
	$[(AGI_{HI}/i)/(1+i)^{x}]$
	)
)	

†x=22z+p; with z is integer and p is rest

We already mentioned that growing willow in incomplete rotation cycles will result in low AGI for willow over these years. Figure 4-3 demonstrates that willow is only economically viable when it is grown more than 15 consecutive years. However, this does not take into account the trade-off that we explained in Figure 4-1 and Figure 4-2, where we indicated the trade-off between high income during remediation and faster remediation. It might indeed be economically justified to grow willow for a couple of years (with a negative AGI) if this can be compensated by the faster introduction of a HI crop on the land.

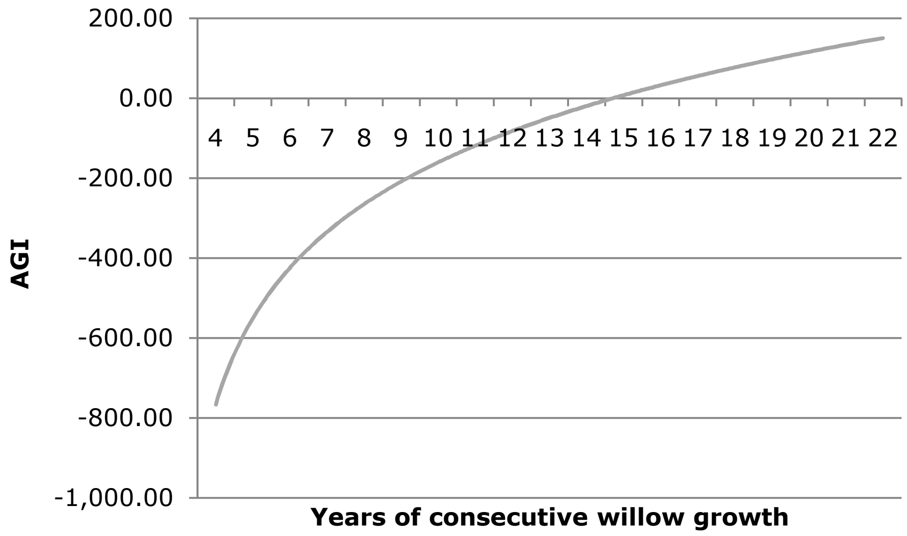


Figure 4-3 The AGI of willow (AGI*) as a function of years of rotation

Table 4-10 R with rapeseed (combined with energy maize) as alternative crop

	C_x
If (	
$C_0 < C_x$;	
$R_{previousC0}$;	
	(
	$[([1-(1/(1+i)^x)]/i) \cdot AGI_{RS}] + [(AGI_{HI}/i)/(1+i)^x]$
	)
)	

4.1.4 Results

4.1.4.1 Base case

Table 4-11 Parameter values in the model

Parameter	*Base case*	*Sensitivity analysis*	
Willow			
P_W (€ ton^{-1} dm)	50	75	100
BP_W (ton dm ha^{-1} y^{-1})	6	8.8	15.6
Harvest and chip	800	600	400
Stool removal	1,500	1,125	750
Cuttings	0.12	0.09	0.06
Cycle	22	25	28
Leaves harvested (g)	0	1	
Energy maize			
External cost	in model	not in model	
P_{EM} (€ ton^{-1} fm)	30	25	
Rapeseed			
BP_{RS}	5.2	8	
Other uses	878	1,050	1,209
pH	5	5.5	
Extraction	linear	non linear	
E_i	Table 3-1,Table 3-2, Table 3-3	Vangronsveld *et al.* (2009)	
Legislation	Food threshold value	Soil standard	
Uncertainty level of food threshold values	10%	5%	

Table 4-12 Results model BASE CASE, indicating for each C_0 the remediating crop that should be used to reach C_q that leads to the highest R (€ ha^{-1}) and r (€ ha^{-1} y^{-1})

C_0	C_q	*Rem. crop*	*Max.* $R_{rem} + R_{HI}$	*r*
12.0-10.1	10 (maize)	EM	26,067-26,611	1,042.68-1,064.44
10.0-4.0	10 (maize)	-	31,500	1,260
3.9-3.5	3.4 (asparagus)	W	32,586-71,505	1,303.44-2,860.20
3.4-0.9	3.4 (asparagus)	-	90,479	3,619.00
0.8-0.7	0.8 (peas fresh)	-	100,258	4,010.32
0.6	0.5 (endive)	W	118,297	4,731.88
0.5-0.0	0.5 (endive)	-	148,158	5,926.32

Table 4-12 shows that for initial contamination levels (C_0) above 4 mg kg^{-1}, plant-based technologies do not benefit from growing HI crops. This is due to the long duration until C_q is reached, which reduces the income from HI crops to a negligible amount. Since grain from maize can be used without any problem when soil concentrations lie below 10 mg kg^{-1}, the model suggests growing energy maize until 10 mg kg^{-1} is reached, after which maize for grain could be grown. Between 10 and 4 mg kg^{-1}, remediation would take too long for any of the alternative crops and the model therefore suggests immediately growing maize for the grain[94]. For C_0 between 3.9 and 3.5 mg kg^{-1}, willow is suggested as remediating crop until C_q = 3.4 mg kg^{-1}, from where asparagus should be grown. With C_0 between 3.4 and 0.9 mg kg^{-1} asparagus should be grown immediately, while with C_0 between 0.8 and 0.7 mg kg^{-1} peas should be grown. The model only suggests to remediate for a concentration in soil of 0.6 mg kg^{-1}. A fast remediation with willow, and the high income of endive from 0.5 mg kg^{-1} compensates for the low income of willow during remediation. Below 0.5 mg kg^{-1} endive can be grown immediately.

4.1.4.2 Alternative crops

<u>Price of willow</u>

For SRC of willow the base case model uses a price of € 50 ton dm^{-1}, based on the price of A-wood in Table 3-15. In this table also other types of wood with their respective prices are given. We don't consider other wood types as these take into account the fact that the wood is treated or contaminated. This fact is in our model taken care of through internalizing the externality of metals. For category A-wood, € 50 is an average (to low) price. There is a lot of import and export of woody biomass waste for energetic valorization and recycling, mostly between Belgium, The Netherlands, Germany, France, and Sweden. OVAM (2010) presents data for 2007 and 2008, showing that the wood market is emerging and that prices will fluctuate in the future, depending on market

[94]The model suggest growing maize for the grain and the rest of the plant material is left on the field as would have been the case in normal circumstances. The model does not suggest energy maize since this crop is harvested in total and would bring with disposal costs for the rest product.

evolutions. Given the European renewable energy goals set for 2020, a shortage in wood and biomass for renewable energy production is expected, leading biomass (and wood) prices to raise. Therefore, we analyzed the effect of the price of willow going up with 50% and 100%, to resp. € 75 and € 100 ton dm^{-1}, *ceteris paribus* (Table 4-19).

Willow cultivation costs

In Flanders, experience with willow as an agricultural crop is scarce. Therefore, one of the goals of the installation of the second plantation on the experimental field is to gain experience with practical aspects of these cultures (weed control, pest control, fertilizer needs, planting, and harvesting techniques). In the future this could reduce costs. We analyzed what the effects are of reducing the costs of the cuttings, chipping and the final stool removal with 25% and 50% (Table 4-13). For the same reasons as with the price change of willow, the impact is small. Willow is not chosen more often, and r only increases lightly, compared to the base case.

Table 4-13 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), base case (chipping = € 800, stool removal = € 1,500, and cuttings = € 0.12) compared with the case where these costs are reduced with 25% and 50%, *ceteris paribus*

C_0	C_q	*Rem. crop*	*Max. R_{rem} + R_{HI}*	*r*
100% (base case)				
3.9-3.5	3.4 (asparagus)	W	32,586-71,505	1,303.44-2,860.20
0.6	0.5 (endive)	W	118,297	4,731.88
75%				
3.9-3.5	3.4 (asparagus)	W	34,556-72,588	1,382.24-2,903.52
0.6	0.5 (endive)	W	119,380	4,775.20
50%				
3.9-3.5	3.4 (asparagus)	W	36,526-73,670	1,461.04-2,946.80
0.6	0.5 (endive)	W	120,462	4,818.48

<u>Willow biomass potential</u>

For willow, the first harvest on the experimental field occurred 3 years after planting. The average biomass productivity over 3 to 4 years was 6 ton dm ha^{-1} year^{-1}. This is low in comparison to the average expected productivity values found in literature (10-12 ton dm ha^{-1} year^{-1}) (Ceulemans *et al.*, 1996; Kopp *et al.*, 2001; Volk *et al.*, 2004). Vangronsveld *et al.* (2009) base calculations on a biomass yield of 10.2 ton dm ha^{-1} year^{-1}. The low biomass yield on the field is not surprising since soil conditions in the area are not very favorable (dry, poor and sandy) for growing these species (Ruttens *et al.*, 2008). It is expected that biomass will increase during the next rotation cycles (Witters *et al.*, 2009). Moreover, there were obvious clonal differences: some clones reached levels of 13 ± 5 ton dm ha^{-1} year^{-1}, and ongoing trials (on the same site in Lommel) with clones that are not yet available on the market (breeding program of the Research Institute for Nature and Forest, INBO, Belgium) showed that a production of 15.6 ton dm ha^{-1} year^{-1} could be reached (unpublished data).

Table 4-14 Number of times (sum of different C_0) that willow is chosen as the alternative crop (willow) and rapeseed or energy maize are chosen (other), given different biomass yields of willow per hectare per year (BP_W), *ceteris paribus*, with an indication of willow contributing in soil management (% of total land cases (n = 121)), and willow contributing to actual remediation (% remediation)

BP_W	C_0	*willow*	*% total*	*other*	*% remediation*
6	3.9-3.5, 0.6	6	5.0%	20	23%
8	4.1-3.5, 0.7-0.6	9	7.4%	20	31%
10	4.3-3.5, 0.9-0.6	13	10.7%	20	39%
12	4.5-3.5, 1.0-0.6	16	13.2%	20	44%
14	4.8-3.5, 1.0-0.6	19	15.7%	20	49%
15.6	10.2-10.1, 5.0-3.5, 1.1-0.6	24	19.8%	18	57%

In the base case (BP_W = 6), willow is used 6 times as a remediation crop. Given the total range of Cd concentration in soil (0-12 mg kg^{-1}), willow is thus used in 5% of cases. However, not for all concentrations the model suggests remediating (due to long remediation periods, the model might suggest growing a HI crop that is possible at higher C_q). The model only suggests actual plant-based technologies in 20 other cases (with energy maize, for soil concentrations between 12 and 10 mg kg^{-1}). Therefore, within the cases where the model suggest remediating before growing a HI crop, willow is suggested in 23%. When BP_W reaches levels as in literature (10-12 ton dm), then willow is chosen in 11-13% of soil management strategies, and in the case actual remediation is suggested, willow is chosen 39%-44% of times. In case the willow clone from the INBO breeding program could be commercialized, willow would be preferred as the remediating crop in 57% of cases.

These positive results can be explained by the fact that changing the biomass potential has an effect on both remediation duration and r_W. In Table 4-15 we calculated the minimum r_W so that willow would be chosen as the management option for Cd contaminated soil in 25% and 50% of cases, for BP_W = 6, 12, and 15.6. In the base case, given BP_W = 6, and r_W = € 110, this percentage was low (5%).

Table 4-15 Minimum r_W (€ ha^{-1} y^{-1}) for willow to be chosen within a soil management strategy in 25% and 50% of cases (n = 121), for BP_W = 6, 12, and 15.6

BP_W	*r_W (25%)*	*r_W (50%)*
6	1,200	1,300
12	1,050	1,250
15.6	900	1,200

Rotation cycles

In literature (Abrahamson *et al.*, 2002; Rosenqvist and Dawson, 2005; Meiresonne *et al.*, 2009), rotation cycles of 7 · 3 + 1 years are suggested which would lead to optimal biomass yields. Alternatively, the model investigates the effect of growing SRC of willow in cycles of 8 · 3+1 years, and 9 · 3+1 years[95], assuming the yearly biomass yield remains unchanged. This will lead to a higher AGI (and r) since the fixed cost of planting and stool removal are now spread over more cycles. For a rotation cycle of 25 years, r_W = € 132.98. Compared to the base (€ 110.26), this is a 21% increase. For a rotation cycle of 28 years, r_W = € 150.49 (compared to the base, this is a 36% increase). There is only a negligible impact of longer rotations. Willow is not preferred as the management strategy in additional cases compared to the base case, but on those lands, average r raises (1%-5%).

[95]Alternatively we could also investigate what the effect is of assuming cycles of 4 to 5 years (Vande Walle, 2007a). However, assuming that biomass yields remain the same over time, this would lead to the same results as assuming more cycles.

Table 4-16 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), base case (rotation cycle 22 years) compared with rotation cycles of 25 and 28 years, *ceteris paribus*

C_0	*C_q*	*Rem. crop*	*Max. R_{rem} + R_{HI}*	*r*
22 (base case)				
3.9-3.5	3.4 (asparagus)	W	32,586-71,505	1,303.44-2,860.20
0.6	0.5 (endive)	W	118,297	4,731.88
25				
3.9-3.5	3.4 (asparagus)	W	32,791-71,505	1,311.64-2,860.20
0.6	0.5 (endive)	W	118,297	4,731.88
28				
3.9-3.5	3.4 (asparagus)	W	34,308-71,505	1,372.32-2,860.20
0.6	0.5 (endive)	W	118,297	4,731.88

<u>Harvest leaves</u>

Table 4-17 gives an overview of what the value of the remaining biomass of crops left on the field would be if collected appropriately. All remainders are combusted to replace cokes-based heat, resulting in a private economic value (B) and an external benefit because of potential CO_2 abatement (A). Final results are calculated by subtracting from the combustion value the cost of collecting, transport, disposal of ashes, and taking into account opportunity costs (*i.e.* alternative applications) of the remainders. In the base case, willow stems are harvested without leaves, which are left on the field. Harvesting leaves could have an impact on the economically most efficient remediation crop-HI crop combination through (i) AGI of willow, as harvesting leaves does not only come with extra costs, but also opens additional energy perspectives, and through (ii) shortened remediation duration, as willow leaves extract up to 40 mg kg^{-1} dm (compared to the stem which extracts 25 mg kg^{-1} dm). Although Table 4-17 gives an overview for the three crops, we only discuss willow leaves[96].

[96]For energy maize, we assume that 2.2 ton dm straw could be collected in addition to the 20 ton dm we already harvested. Generally, rapeseed seeds are harvested, while the straw is used as hay. Rapeseed straw could be combusted alternatively.

Table 4-17 Remaining remediating crop biomass on the field: valorization of combustion of remainders with subsequent landfill of ashes based on the impact on CO_2 abatement (A) and on market value (B)

		Energy maize straw	*Rapeseed straw*	*Willow leaves*
Harvest				
Biomass (ton dm ha^{-1})		2.2	2.2	1.2
Biomass (ton fm ha^{-1})		2.2	2.2	4 (30 dm%)
Economic reference value (€ ha^{-1})	(1)	0	111 (hay)	0
Collecting (A) (€ ha^{-1})	(2a)	0.78†	0	0.78†
Collecting (B) (€ ha^{-1})	(2b)	15.73‡	0	15.73‡
Transport (A) (€ ha^{-1})§	(3a)	0.13	0	0.23
Transport (B) (€ ha^{-1})§	(3b)	4.4	0	8
Total extra harvest (A)	(4a)	0.91	0	1.01
Total extra harvest (B)‡	(4b)	20.13	0	23.73
Additional harvest-worst case	(4')	150	0	150
Combustion				
GJ ton^{-1} dm¶		16-19	18.1	20
GJ ha^{-1}		35.2	39.82	24
Ton CO_2 ha^{-1}††		4.71	5.33	3.21
Combustion (A) (€ ha^{-1}) ‡‡	(5a)	94.16	106.52	64.20
Combustion (B) (€ ha^{-1}) ‡‡	(5b)	109.24	123.58	74.48
Landfill of ashes				
Ash content (m%)¶		3-5	4	4-5
Ton ashes ha^{-1}		0.07	0.09	0.05
Cat. 1 disposal (€ ha^{-1}) #	(6')	2.77	3.70	2.02
Cat. 2 disposal (€ ha^{-1}) #	(6)	1.06	1.41	0.77
Marginal valorization				
B (min) (€ ha^{-1})	(7')	(63.66)	8.88	(101.27)
(5b)-(6')-(4b)-(4')-(1)				
B (max) (€ ha^{-1})	(7)	88.05	11.17	49.98
(5b)–(6)–(4b)-(1)				
A (5a)-(4a) (€ ha^{-1})	(8)	93.25	106.52	63.19

†12.83 liter diesel ha^{-1} (Borjesson, 1996), 40.91 MJ l^{-1}, 74.1 kg CO_2 GJ^{-1}; ‡€ 1.226 l^{-1} (in 2010); §1.3 MJ ton^{-1} fm km^{-1} (Borjesson, 1996), 30 km, € 2 ton^{-1} fm; ¶www.ecn.nl and Borjesson (1996); ; #cat. 1: € 42 ton^{-1}, cat. 2: € 16 ton^{-1}; ††Cokes: 107 kg CO_2 GJ^{-1} · 80% boiler efficiency = 134 kg CO_2 GJ^{-1}; ‡‡Cokes: € 0.09 kg^{-1}, 29 GJ ton^{-1}, 80% boiler efficiency

The economic reference value (1) in Table 4-17 is the value of straw and leaves in regular, uncontaminated circumstances. Left on the field, maize straw and willow leaves have a value as organic material. However, the farmer will apply the maximum amount of fertilizer on his land anyway (reducing the economic value of straw and leaves to 0). We also neglect the impact on soil fertility of leaving this straw and leaves on the field. Therefore, in the reference case, the value of leaves from SRC and straw from energy maize is € 0. Rapeseed straw is collected in regular circumstances and used as hay with a value of € 111 ha^{-1}, including collecting and transport (<20 km).

Additional collection and transport of leaves and straw results in CO_2 emissions (row 2a and 3a) and also has a private impact (row 2b and 3b). Transport costs per ton are determined as before (Eq. 7). The remaining products on the field are transported to a combustion installation (30 km), resulting in a cost of € 2 per ton. In row (4') we include a worst case scenario where in addition to all other costs, extra harvest costs of the remainders are added (€ 150 ha^{-1}). For rapeseed, the value in rows 2a - 4 is 0 because straw was harvested before as well, there is no marginal effect.

Row 5a and 5b show the economic value of combusting the remainders, based on respectively CO_2 abatement and market price. Row 6 and 6' represent disposal costs of ashes after combustion, with cat. 1 disposal being more expensive than cat. 2 disposal. The economic impact of the combustion of remainders instead of their current use is then calculated as the economic value of combustion minus the costs of ash disposal, transport, collection, additional harvest, and minus the economic reference value, based on market prices (row 7 and 7') and based on CO_2 abatement (row 8). Minimum and maximum values (row 7' and 7) differ because of (i) disposal categories and (ii) additional harvest costs of € 150 ha^{-1} (row 4').

Only for rapeseed straw, the effect of the alternative use is always positive, even in the worst case (*i.e.* in the case of additional harvest costs of € 150 ha^{-1}). For energy maize straw and willow leaves, the effect is negative when additional harvest costs are added.

The cost/benefit of harvesting leaves (row 7 in the best case, and row 7' in the worst case) is introduced into the model to analyze whether it is economically efficient to remove the leaves. Harvesting leaves also has an impact on remediation duration.

Table 4-18 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), base case (leaves on the field, g=0) compared with leaves harvested (g=1), in a worst case and best case scenario, *ceteris paribus*

C_0	C_q	*Rem. crop*	*Max.* $R_{rem} + R_{HI}$	*r*
g=0 (base)				
3.9-3.5	3.4 (asparagus)	W	32,586-71,505	1,303.44-2,860.20
0.6	0.5 (endive)	W	118,297	4,731.88
g= 1 (worst)				
4.0-3.5	3.4 (asparagus)	W	37,046-75,044	1,481.84-3,001.76
0.7-0.6	0.5 (endive)	W	108,864-124,718	4,354.56-4,988.72
g = 1 (best)				
4.1-3.5	3.4 (asparagus)	W	33,308-75,044	1,332.32-3,001.76
0.7-0.6	0.5 (endive)	W	108,864-124,718	4,354.56-4,988.72

worst = worst case: additional harvest costs for the leaves of € 150 ha^{-1}, *best* = best case: no additional harvest costs

Table 4-18 demonstrates the impact of harvesting leaves with additional (worst case) and without additional harvest costs for leaves. Willow is not only chosen in more cases, the AGI is also raised for each of the zones. In these zones, the effect on AGI lies between € 72 and € 399 in the best case, and between € 0 and € 377 in the worst case. If we assume that the zones with different Cd concentrations are uniformly distributed over the region, the average augmentation in R and r are respectively € 16 ha^{-1} and € 31 ha^{-1} $year^{-1}$.

Table 4-19 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), base case (willow price, P_W = € 50) compared with P_w = € 75 and P_W = € 100, *ceteris paribus*

C_0	C_q	*Rem. crop*	*Max.* $R_{rem} + R_{HI}$	*r*
€ 50 (base)				
3.9-3.5	3.4 (asparagus)	W	32,586-71,505	1,303.44-2,860.20
0.6	0.5 (endive)	W	118,297	4,731.88
€ 75				
3.9-3.5	3.4 (asparagus)	W	34,459-72,110	1,378.36-2,884.20
0.6	0.5 (endive)	W	118,902	4,756.08
€ 100				
3.9-3.5	3.4 (asparagus)	W	36,332-72,714	1,453.28-2,908.56
0.6	0.5 (endive)	W	119,507	4,780.28

Both price changes induce no large effect on economically efficient crop choices, which is very obvious since r_W is € 222 and € 334 respectively when the price per ton dm is € 75 and € 100. These numbers have to be compared with r_{EM} and r_{RS} of respectively € 1,043 and € 878.

<u>Alternative conversion of rapeseed</u>

The phytoremediation model is based on selling rapeseed as such to a biodiesel producer. The model also foresees other conversion options for rapeseed. When rapeseed is pressed to PPO on the farm, two options exist: PPO is used for personal use, or for farming purposes (tractor). Both options have a higher AGI' than energy maize, and rapeseed therefore becomes an alternative option (Table 4-20). Remember from (3.2.2.2) that this PPO has to fulfill certain requirements to be allowed to be sold on the Belgian market. This has to be taken into account when deciding on using rapeseed for PPO. Since remediation goes slow for both energy maize and rapeseed, they both have no advantage of growing HI crops, except for C_0 close to 10 mg kg^{-1}. Therefore, we can simplify and state that rapeseed is chosen when r_{RS} is equal to r_{EM}. Taking into account rotation of rapeseed with energy maize, and external benefits and costs (which are zero for rapeseed), this translates into an AGI_{RS} = € 1,042.68 (equal to $AGI'_{EM} = r_{EM}$). In general, the AGI of rapeseed can be an amount lower for rapeseed than for energy maize, equal to the external cost of metal waste disposal related to energy maize, for rapeseed to be chosen as the soil management option.

Table 4-20 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), base case (rapeseed for biodiesel) compared to alternative conversion of rapeseed (personal use (r_{RS} = € 1,209) and use on the farm (r_{RS} = € 1,050)), *ceteris paribus*

C_0	C_q	*Rem. crop*	*Max.* $R_{rem} + R_{HI}$	*r*
r_{RS}= € 878				
12.0-10.1	10 (maize)	EM	26,067-26,611	1,042.68-1,064.44
r_{RS}=€ 1,209				
12.0-10.1	10 (maize)	RS	30,236-30,329	1,213.04-1,213.16
r_{RS}=€ 1,050				
12.0-10.1	10 (maize)	RS	26,250-26,635	1,050.00-1,065.40

Biomass potential of rapeseed

Based on Vangronsveld *et al.* (2009), we use a biomass yield of 8 ton dm ha^{-1}. This is more than 50% higher than the average found in literature, which we used in the base case. This difference could be explained by the fact that the remediation crops in Vangronsveld *et al.* (2009) were grown immediately after the last agricultural activity on the field. The alternative crops could therefore benefit from the nutrients still present in the soil due to fertilization by farmers.

Due to the increase in BP_{RS}, rapeseed accumulates the same amount of Cd as energy maize (Eq. 21). Therefore, the choice between energy maize and rapeseed solely depends on income. When rapeseed is sold as such, r_{RS} = € 1,022, when rapeseed is pressed to PPO and used for a personal vehicle, r_{RS}= € 1,520, whereas when PPO is used on the farm, r_{RS} = € 1,274. Rapeseed will be preferred over energy maize when r_{RS} > € 1,042.68. Moreover, when r_{RS} > € 1,260 (= AGI_{maize}) rapeseed will be grown instead of maize from 10 mg kg^{-1}.

Table 4-21 Results model, when r_{RS} > AGI_{maize}

C_0	*C_q*	*Rem. crop*
12.0-6.5	differs	RS
6.4-4.0	3.4 (asparagus)	RS
3.9-3.5	3.4 (asparagus)	W
3.4-0.9	3.4 (asparagus)	-
0.8-0.7	0.8 (peas fresh)	-
0.6	0.5 (endive)	W
0.5-0.0	0.5 (endive)	-

Price energy maize

In the base case we assume a maize price of € 30 ton^{-1} fm. Reducing the price of energy maize with 15% to € 25 ton^{-1} fm reduces AGI_{EM} to € 960 and r_{EM} to € 743. Moreover, AGI of maize for grain (allowed after remediation) reduces from € 1,260 ha^{-1} year^{-1} to € 960 ha^{-1} year^{-1}. This does not change plant-based technology strategies (Table 4-22). But it could have an effect on welfare, since for each C_0 r is reduced with at least 5%.

Table 4-22 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), price energy maize = € 25, *ceteris paribus*

C_0	*C_q*	*Rem. crop*	*Max. R_{rem} + R_{HI}*	*r*
12.0-10.1	10 (maize)	EM	18,567-19,111	742.68-764.44
10.0-4.1	10 (maize)	-	24,000	960
4.0-3.5	3.4 (asparagus)	W	25,677-66,942	1,027.08-2,677.68
3.4-0.9	3.4 (asparagus)	-	84,854	3,394.16
0.8-0.7	0.8 (peas fresh)	-	94,008	3,760.32
0.6	0.5 (endive)	W	114,241	4,569.64
0.5-0.0	0.5 (endive)	-	143,158	5,726.32

External cost of metal in rest product

If we assume that the external cost of metal in the rest product, *i.e.* the cost to guarantee that the metals do not end up in the environment again and cause environmental and/or health problems, are already taken into account through the biomass price, we have to take them out of the model. This would have the following implications. Between 12 mg kg^{-1} and 6.5 mg kg^{-1}, it would not matter whether you would grow energy maize or just maize, since the income for both crops are now the same (no external cost for energy maize). Between 6.5 and 4.0 mg kg^{-1}, energy maize would be grown, and from 3.9 until 3.5 mg kg^{-1} willow is grown, both followed by asparagus. From then, things remain the same as in the base case.

4.1.4.3 pH

Alternative crops

For the first experimental planting (2004), soil pH-KCl ranged between 4.7 and 6.0, with blocs defined as low pH blocs (4.7-5.0) and high pH blocs (5.1-6.0). Before the second field of SRC of willow, and rapeseed and energy maize was set up on the experimental site, the whole field was limed one month before the realization of the plantation. As a result of the liming, the pH had shifted to a range of 5.6-6.7. For SRC of willow (and poplar) Ruttens *et al.* (2008) conclude that the installation of a SRC plantation on contaminated soils is only realistic in conditions of sufficiently high soil pH. In cases of low pH, the field should be limed prior to the planting. The optimal pH may be dependent on the observed metal availability. For rapeseed, there were no data available yet. For energy maize, metal concentrations in the different plant organs were always higher in the plot with low pH than in the plot with high pH (Ruttens *et al.*, 2008). We based our calculations on average metal concentrations, since there were no data available yet on the relationship between metal concentration and soil pH for the remediating crops.

HI crops

For HI crops, this relationship has already been studied in detail. The relation between soil pH, Cd concentrations in soil and Cd concentrations in harvested product is not a straightforward relation. At low concentrations of Cd in soil, low as well as high concentrations of Cd are found in crops. Moreover, treatments such as peeling and boiling can cause changes in Cd concentrations in crops.

Table 4-1 shows for each HI crop the impact of pH on the maximum allowed level of Cd in soil, for the HI crop grown on this land still to comply with food- and fodder threshold values. The magnitude of the impact of pH on the allowed soil contamination level differs between HI crops, with a smaller impact when allowed concentrations are smaller, compare *e.g.* beans and spinach. Since remediation is linear and will not slow down at small concentrations in soil, one could expect an effect of pH on the economically most efficient combination of remediation crop with HI crop.

Table 4-23 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), pH = 5.5, *ceteris paribus*

C_0	C_q	*Rem. crop*	*Max* $R_{rem} + R_{HI}$	*r*
12-10.1	10 (maize)	EM	26,067-26,611	1,042.68-1,064,44
10.0-4.2	10 (maize)	-	31,500	1,260
4.1-3.9	3.8 (beans)	W	35,968-51,153	1,438.72-2,046.12
3.8-3.6	3.8 (beans)	-	65,392	2,615.68
3.5	3.4 (asparagus)	W	71,505	2,860.20
3.4-1.2	3.4 (asparagus)	-	90,479	3,619.16
1.1-1.0	0.9 (endive)	W	97,260-118,297	3,890.4-4,731.88
0.9-0.0	0.9 (endive)	-	148,158	5,926.32

Figure 4-4 compares Table 4-12 and Table 4-23 and demonstrates that it pays off for farmers to lime their lands (pH raises) when they are situated in a region with concentration between 4.1 and 3.6 mg kg^{-1} soil, and between 1.1 and 0.6 mg kg^{-1} soil, since a higher pH results in other and more potential HI crops (Table 4-1), resulting in a higher R (and r).

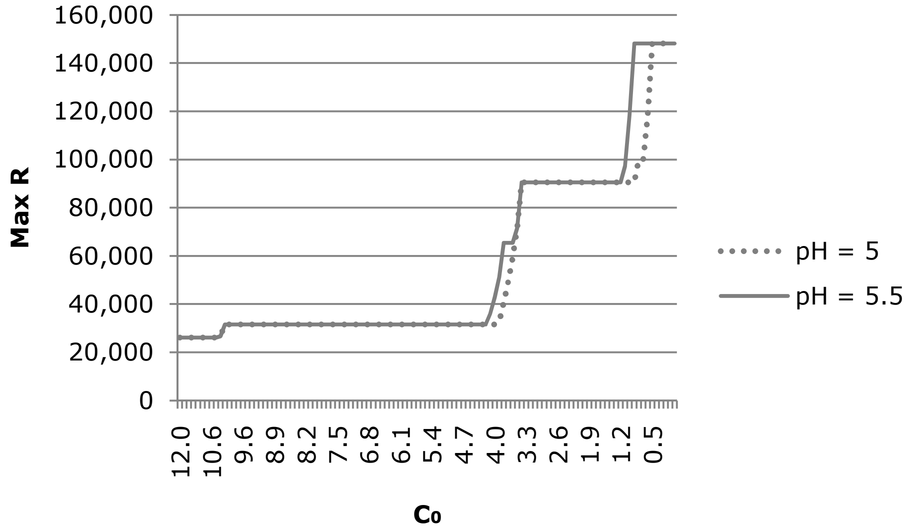


Figure 4-4 Comparison of maximum R (€ ha^{-1}) over infinity (i = 4%) based on pH = 5, and pH = 5.5, *ceteris paribus*, starting from C_0

However, the liming will bring with additional costs. Therefore, we calculated the maximum cost of liming per hectare per year as the additional r due to liming (Table 4-24). These maximum liming costs vary between € 179 per hectare per year for more heavily Cd contaminated land and € 2,307 per hectare per year for lightly Cd enriched land.

Table 4-24 Maximum liming cost in € per hectare per year (Δ r) for R to be the same for pH = 5.5 as for pH = 5

C_0	*R (pH = 5)*	*R (pH = 5.5)*	*Δ R*	*Δ r*
4.1	31,500	35,968	4,468	179
4.0	31,500	42,788	11,288	452
3.9	32,586	51,153	18,567	743
3.8	40,790	65,392	24,602	984
3.7	49,363	65,392	16,029	641
3.6	59,299	65,392	6,092	244
1.1	90,479	97,260	6,780	271
1.0	90,479	118,297	27,818	1,113
0.9	90,479	148,158	57,679	2,307
0.8	100,258	148,158	47,900	1,916
0.7	100,258	148,158	47,900	1,916
0.6	118,297	148,158	29,861	1,194

Δ R = the increment in R compared to R in the base case, Δ r = the increment in r compared to the base case

4.1.4.4 Non linearity of metal accumulation

We present an overview if long-term data on the relation between $E_{i(t+1)}$ and C_{it} were available. Instead of using a constant uptake approach to calculate the remediation period, a constant fading approach is used (Figure 4-5). From Japenga *et al.* (2007) we derive the following equations:

$REM_{it} = A \cdot BP_i \cdot E_{it}$ (Eq. 27)

$Q_t = Q_{t-1} - REM_{it}$ (Eq. 28)

$C_{it} = Q_t / (A \cdot d_i \cdot \rho)$ (Eq. 29)

$\text{Log } E_{i(t+1)} = X + Y \cdot \log C_{it}$ (Eq. 30)

REM_{i1} is the amount of metals removed from the soil after the first year by growing crop i with a first year extraction potential E_{i1}. Q_1 represents the total amount of metals in soil after the first harvest. The metal concentration in soil after the first harvest is then recalculated as in (Eq. 29). If $C_{i1} > C_q$, then E_{i2} (*i.e.* the metal extraction by the plant given the new metal concentration in the soil C_{i1}) is calculated (Eq. 30). The difference with the approach in (Eq. 19)-(Eq. 21) thus lies in the determination of X and Y. X and Y depend not only on plant type, but also on soil type. Japenga *et al.* (2007) have determined X and Y for Alpine Pennycress (*Thlaspi caerulescens)* and Hartweg's Lupine (*Lupinus hartwegii)*. Koopmans *et al.* (2008) calculated X and Y for Alpine Pennycress and Perennial Ryegrass (*Lolium perenne L.*).

The extraction potential of Cd which we use for willow, *i.e.* 25 mg kg^{-1} (stem) is measured at an average concentration in soil of 7 mg kg^{-1}. If we take Y = 0.98 (based on values found in literature, but in fact this is just an example, since long term data for willow were not found), we arrive at a value for X of 0.57 (Eq. 30). We can then determine the remediation duration as a function of initial Cd concentration (C_0). Figure 4-5 represents this relation for C_q = 1.2 mg kg^{-1}.

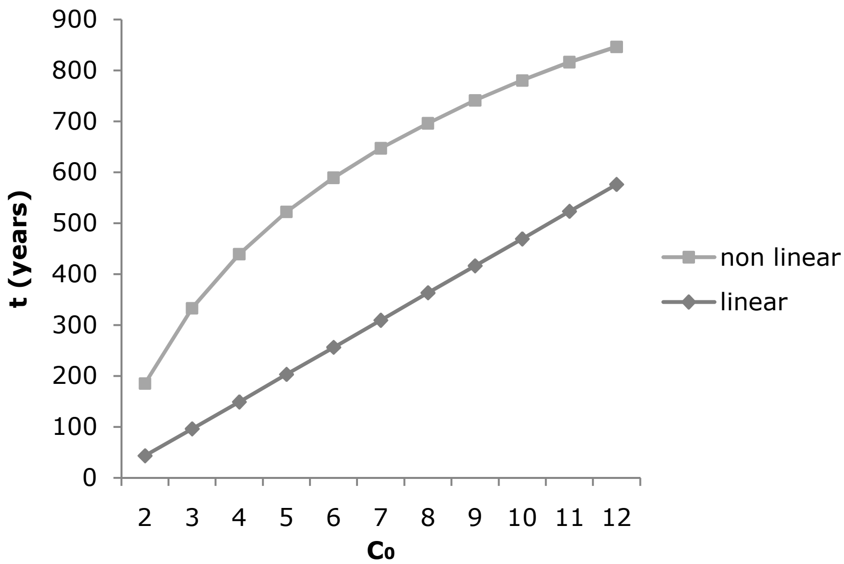


Figure 4-5 Remediation duration (t in years) by harvesting willow stems for different begin concentrations (C_0) to reach C_q (1.2 mg kg^{-1}), given X=0.57, and Y=0.98

Obviously, linear accumulation results in an underestimation of remediation duration, since in the non linear case extraction is a function of the concentration of metals in soil. Applying (Eq. 30), and fixing C_0 at 7 mg kg^{-1},

Figure 4-6 shows that over time, E_{it} (where i indicates a certain crop and t a certain point in time) reduces exponentially.

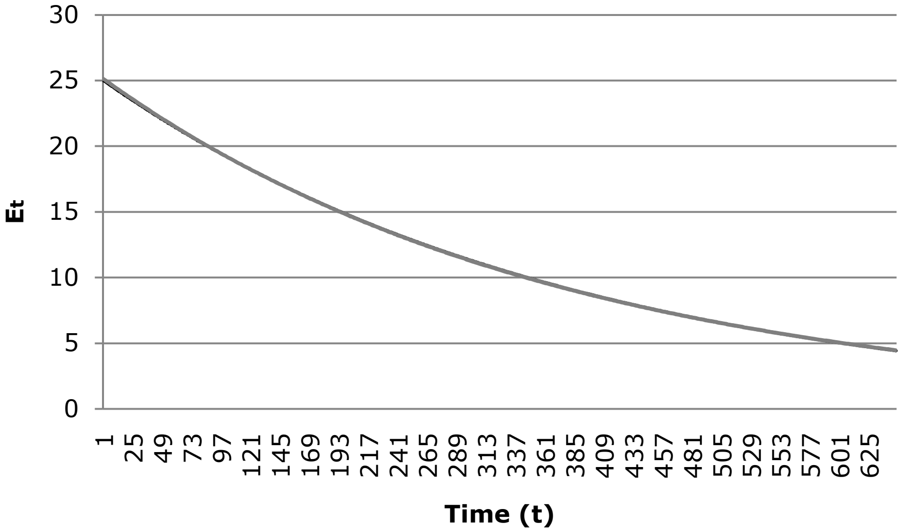


Figure 4-6 Extraction (E_t) of Cd by willow stems at time t, assuming C_0=7 and C_q = 1.2, given Y=0.98 (and the resulting X=0.57)

Also other authors attempt to cover the non linear extraction of metals. For willow, Maxted *et al.* (2007a) represent the Cd uptake by the stems (mg kg^{-1}) as an asymptotic function of total Cd content in soil (mg kg^{-1}), based on average literature data: Cd_{stem} = 1.87 Cd_{soil}/ (1 + 0.194 Cd_{soil}). Maxted *et al.* (2007b) model the extraction of Cd and Zn from soil for different pH levels.

4.1.4.5 Extraction potential

Table 4-25 Extraction of Cd for energy maize, rapeseed, and willow, based on base case values and Vangronsveld *et al.* (2009)

	Base case			*Vangronsveld et al. (2009)*	
	Biomass	*Cd*	*Cd removal*	*Cd*	*Cd removal*
	(ton dm ha^{-1})	*(mg kg^{-1} dm)*	*(kg ha^{-1} y^{-1})*	*(mg kg^{-1} dm)*	*(kg ha^{-1} y^{-1})*
EM	20	1.08	0.022	3 (+178%)	0.06
RS	5.2	2.68	0.014	6 (+123%)	0.031
SRC-stem	4.8	25	0.120	24 (-4%)	0.12
SRC-leaves	1.2	40	0.048	60 (+50%)	0.072

EM= energy maize, RS= rapeseed, SRC = SRC of willow

Based on Vangronsveld *et al*. (2009) we recalculated Cd removal for each of the crops (Table 4-25). The differences in Cd accumulation between the base case and Vangronsveld *et al.* (2009) could be explained by the fact that data in the base case are based on an average of different species/clones, whereas data in Vangronsveld *et al.* (2009) are based on one specific species/clone. Moreover, accumulation of trace elements is also vulnerable to seasonal conditions, which differ over several years. This results in the following optimal management options for C_0. Compared to the base case, energy maize is chosen once more as the management strategy (C_0 = 3.9 mg kg^{-1}). The impact of faster remediation is only felt on R (and thus on r) when the management option consists of actually applying plant-based technologies (Table 4-26).

Table 4-26 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), E_i from Vangronsveld *et al.* (2009), *ceteris paribus*

C_0	C_q	*Rem. crop*	*Max.* $R_{rem} + R_{HI}$
12.0-10.1	10 (maize)	EM	26,068-28,420
10.0-4.0	10 (maize)	-	31,500
3.9	3.4 (asparagus)	EM	34,020
3.8-3.5	3.4 (asparagus)	W	39,440-70,868
3.4-0.9	3.4 (asparagus)	-	90,479
0.8-0.7	0.8 (peas fresh)	-	100,258
0.6	0.5 (endive)	W	117,254
0.5-0.0	0.5 (endive)	-	148,158

4.1.5 Discussion and conclusion

4.1.5.1 Model outcomes based on food- and fodder threshold values

From the model we learn that in general energy maize could be grown as an alternative and safe crop for ranges of Cd concentration in soil between 12 and 10.1 mg kg^{-1}, after which maize for grain (!) could be grown (maximum allowed soil concentration is 10 mg kg^{-1}). Willow is chosen between 3.9 and 3.5 mg kg^{-1} until 3.4 mg kg^{-1}, after which asparagus can be grown, and from 0.6 mg kg^{-1} to remediate until 0.5 mg kg^{-1} from which moment endive is grown. For a HI crop to be chosen by the model, the HI crop should represent a good combination of high AGI and easy to reach. In the base case asparagus (AGI = € 3,619 and C_q = 3.4) is preferred over beans (AGI = € 2,616 and C_q = 2.7). However, when pH is raised to 5.5 (in the base case the pH = 5), metals become less available and C_q for all HI crops raise, resulting in beans to be preferred over asparagus under certain conditions.

Rapeseed is in competition with energy maize, and our analyses show that energy maize and rapeseed are equally preferred when $AGI_{RS} = r_{EM}$. Energy maize is the preferred crop for large distances to target (DTT), *i.e.* the difference between C_q and C_0, while willow is preferred for average to small DTT. Changing the price of willow (*ceteris paribus*), or changing the costs of willow (*ceteris paribus*) has no effect.

Given the contamination range (12-0 mg kg^{-1}), the model only suggests growing one of the three crops in the context of risk reduction in 15% of cases. Main reasons for this are the fact that it is theoretically safe to grow maize for the grain (!) from 10 mg kg^{-1}, and the fact that, when C_0 already lies below C_q for a HI crop, the model often does not suggest growing one of the three alternative crops (instead, the model suggests immediately safely growing the allowed HI crop). In Table 4-14 we changed BP_W and noticed that a raise in BP_W results in willow to be chosen more often as the remediation crop, and that more often actual remediation is suggested by the model (Figure 4-7). To induce actual remediation (*i.e.* chosen as the preferred option in 50% of cases), the AGI of willow should at least be € 1,200. Our analyses show that this could not be reached by changing parameters separately, but a simultaneous increase in allowed rotation cycles, an increase in biomass yield, a reduction in costs, and the harvest of leaves might be able to realize this.

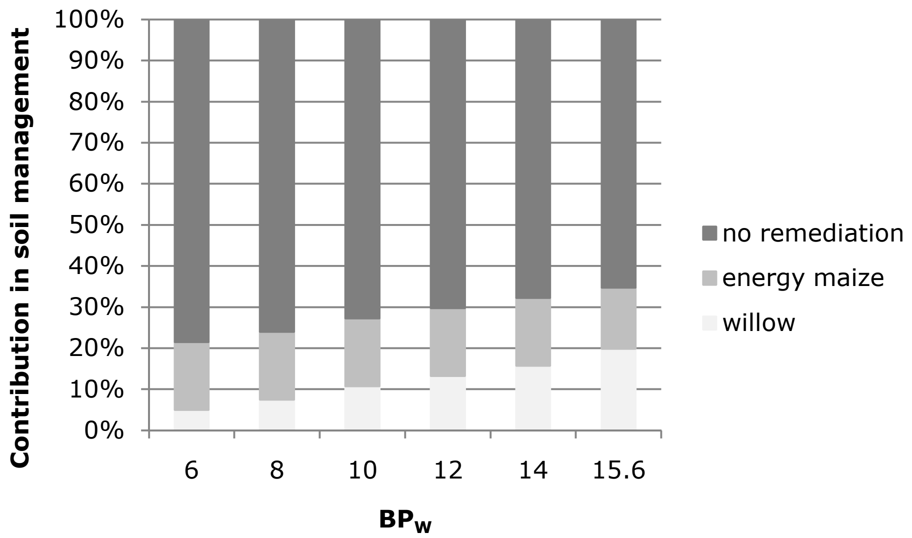


Figure 4-7 Percentage of cases in which willow, and energy maize are chosen as alternative safe crop, and when the model suggest not to grow an alternative crop prior to an HI crop, for different biomass yields of willow

These are results from the model based on food- and fodder threshold values. However, soil standards exist simultaneously. Given $C_0 > 2$ mg kg^{-1}, which is the BSN above which soil should be remediated according to soil standards, the model never suggests to remediate soil until $C_x < 2$ mg kg^{-1}, simply because this is not the economically optimal solution.

4.1.5.2 Legislation on soil remediation: inconsistent?

Control on application of legislation is inconsistent. First, based on food threshold values, some harvests have been confiscated in the past (*e.g.* carrots), but the practice of strictly applying the law is not consistent. Second, crops from farmers in the region are subject to more thorough control than farmers in other regions, even though soil samples tested for soil standards might indicate that the former farmers practice in a safe zone.

The system of control appears to be a trapped system, where a first check is based on soil standards, resulting in a safe and unsafe farming zone (classification made by authors). Soil samples above the accepted soil standard result in crops from unsafe zones being tested thoroughly regarding food threshold values. However, threshold values for some food crops even lie below soil background values for Cd, and while farmers growing crops in a safe zone are allowed to grow these crops, farmers from the unsafe zone see their harvests being confiscated, which is unfair (Table 4-27).

Table 4-27 Food and fodder threshold values (presented as the maximum concentration of Cd in soil (mg kg^{-1} dm)) that lie below soil standards (< BSN, TV, and BV) so that no more than 10% of samples exceed food threshold values after harvest, for pH = 4.5, 5, and 5.5†

Soil standard	*HI crops*	*pH = 4.5*	*pH = 5*	*pH = 5.5*
BSN	Cabbage	-	1.7	1.7
	Oignon	-	1.8	1.8
Target value (TV)	Maize (total)	0.8	1.15	1.4
	Peas	0.8	0.8	0.8
Background value (BV)	Endive	0.3	0.5	0.9
	Celeriac	0.2	0.2	0.3
	Scorzonera	<0.1	<0.1	0.1
	Lettuce	0.5	0.6	0.8
	Spinach	0.3	0.4	0.4
	Carrots	0.3	0.3	0.5
	Celery	<0.1	0.1	0.2
	Leek	0.2	0.3	0.5

†TV(pH)=1.2 · 10 $^{(-0.17 \cdot (5\text{-pH}))}$, results in TV(4.5)=0.99 and TV(5.5)=1.46; BSN(pH)=2 · 10 $^{(-0.17 \cdot (5\text{-pH}))}$, results in BSN(4.5)=1.64 and BSN(5.5)=2.43, BV = 0.7 mg kg^{-1} (VLAREBO)

Table 4-27 should be interpreted as a summary of maximum allowed soil concentrations for crops at different levels of pH to comply with food- and fodder threshold values. The first column of this table however indicates the inconsistency with different soil standards as these lie above the calculated maximum concentrations. *E.g.* following food threshold values, and given a pH of 5, spinach is allowed to be grown at a soil concentration of 0.4 mg Cd kg^{-1}

soil. The first column then indicates that this value lies even below background values (BV) of CD in soil.

Table 4-27 suggests that the current trapped system of dividing soils up into two classes based on BSN (for pH = 5, below 2 mg kg^{-1} is safe, and above 2 mg Cd kg^{-1} soil is unsafe) might not be precise enough to cover all food threshold values. Several vegetables slip through the mazes. Following soil standards, the safe zone (< 2 mg kg^{-1}) allows all crops. However, cabbage and oignon can only safely be grown below soil concentrations of 1.7 and 1.8 mg Cd kg^{-1} soil respectively, according to food threshold values. Moreover, all other crops in Table 4-27 are only safely grown for soil concentrations below target- and background values, when we convert food threshold values in soil concentrations. This might indicate that soils should be divided into more classes. A third class might be introduced based on the background value, resulting in a safe (below BV), relatively safe (below BSN), and unsafe (above BSN) class. Figure 4-8 should be read as follows: when Cd concentration in soil < BV, all crops in the blue, orange and red zone are allowed. For Cd concentrations ≥ BV, but < BSN, crops from the orange and red zone are allowed, whereas for Cd concentration in soil ≥ BSN, crops from the red zone are allowed. Crops from the green region should be dealt with very carefully, since these demand very low Cd concentrations in soil.

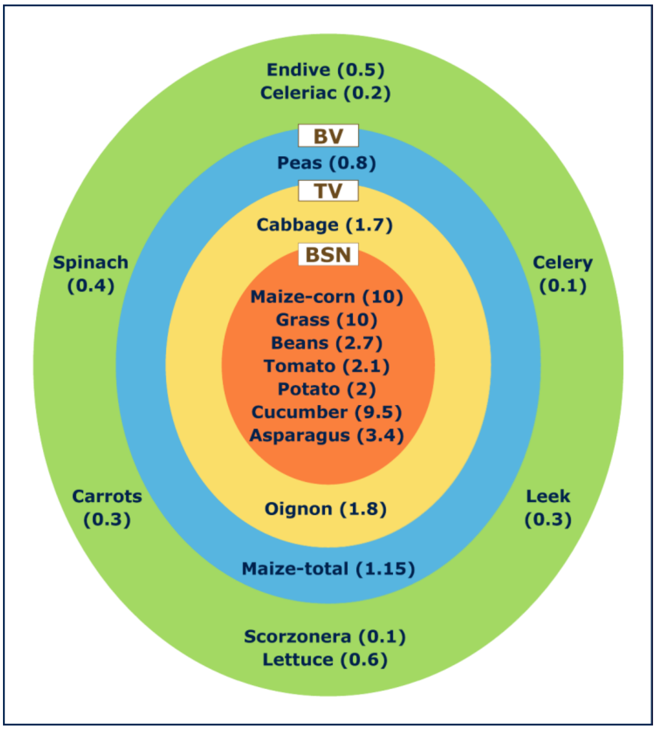


Figure 4-8 Safe (<BV), relatively safe (<BSN), and unsafe zone (>BSN) with an indication of crops allowed in these zones (based on food- and fodder threshold values)

Strictly interpreting current food threshold values, crops only allowed at soil concentrations below BV shouldn't be grown on most Belgian (even uncontaminated sandy) soils. This is already the case for some vegetables in the Campine region[97], but in an inconsistent way. It might however be more appropriate to reconsider the general BV of 0.7 mg Cd kg^{-1} for all soils and make it specific for different soil textures, such as in Table 4-28, which represents adjusted BV's. Doing this will lead to the exclusion of scorzonera and celery only, on Belgian sandy soils (as these crops require soil concentrations of Cd below 0.13 mg Cd kg^{-1} soil, the average BV of Cd in sandy soil in Belgium).

[97]At the moment, the vegetable sector (industry) does no longer conclude contracts in Flemish communities where the possibility of surpassing food threshold values is high.

4.1.5.3 Legislation on soil remediation: inefficient?

It has been recognized for long that uptake by plants is a potential pathway of metal transfer to the human food chain and that a correct quantification critically affects the outcome of environmental risk assessment (Tack *et al.*, 1996). In Belgium, food- and fodder threshold values, and soil standards exist simultaneously to evaluate risks of (Cd) contaminated agricultural soils. Both are based on a human toxicological risk assessment of contaminants. Soil standards are easier in use, since they are averaged to one overall level, based on an average consumption pattern of crops. The basis for the risk assessment is the European precautionary principle, providing adequate safety margins for human and/or ecological health. However, it lacks transparency on the perception of risk as explained below. Theoretically, the cut-off value is determined through a process described later.

De Temmerman *et al.* (1984) estimate the average and upper limit of normal Cd concentrations in non polluted sandy soils at respectively 0.1-0.5 µg g^{-1} and 1 µg g^{-1}. Bierkens *et al.* (2009) refer to this study as the basis for background values for the solid part of soil: 90 percentile values of concentrations in the top soil layer (0-20 cm) of soils in uncontaminated regions. For Cd, this value is equal to the detectable limit by standard measurement techniques. However, in their revision of Cd standards, Bierkens *et al.* (2010) refer to a study performed by Martens *et al.* (1994), of which Table 4-28 is an excerpt and on which background values are based. Tack *et al.* (1997) also determine soil quality reference values based on expected total contents and upper 90% confidence limits of trace elements of observations of (mostly agricultural) unpolluted soils as a function of clay and organic content. In summary, based on De Temmerman *et al.* (1984) normal values of Cd in sandy soils are 0.1-0.5 mg kg^{-1}, according to Martens *et al.* (1994) normal values are 0.04-0.57 mg kg^{-1}, and this has resulted in a background value of 0.7 mg kg^{-1}.

Table 4-28 Normal values for Cd in mg kg^{-1} soil, per texture class, found in Belgian soil, on which background values (Soil Decree) are based

Texture	*Sand*	*Loamy sand*	*Light sandloam*	*Sandloam*	*Loam*	*Clay*	*Heavy clay*
Average	0.13	0.31	0.39	0.41	0.33	1.47	1.19
Max	0.57	1.24	1.14	1.18	0.70	3.37	2.29
Min	0.04	0.04	0.04	0.04	0.02	0.06	0.11

Source: Bierkens *et al.* (2010), based on Martens *et al.* (1994)

BSN are determined twice as high as the upper limit (1 µg g^{-1} soil) found by De Temmerman *et al*. (1984). The Flemish Soil Decree states that BSN correspond to a level of soil contamination which entails a considerable risk of harmful effects for man or environment, taking into account soil characteristics and -functions. Soil remediation standards for different land use types, with human protection as a priority, are based on a risk assessment. In order to perform a risk assessment, the transport of the substance in the environment and of human contact with the substance is quantified (OVAM, 2004a). The concept of risk, which the Soil Decree talks about, is based on a human health risk assessment in which concentrations of contaminants in soil are compared with generic guideline concentrations, which not only use general assumptions about the conditions of the site and the receptors, but are also very cautious (Clarinet, 2002a).

These BSN (for Cd) represent one general threshold value, for all agricultural land (yet adaptable for pH). The procedure for the elaboration of current BSN started from the relation between Cd concentration in plant and Cd concentration in soil, which was averaged based on the relative part of each of these food crops in a typical Belgian food basket[98] (consumption correction factor) (Bierkens *et al.*, 2010), based on Dejonckheere *et al.* (1996). In a next phase, BSN were calculated through the Vlier Humaan model, which also takes into account other transport ways of Cd intake besides food. In general, BSN are based on the most stringent outcome of a human toxicological and eco-toxicological analysis, but the latter is not systematically included in deriving

[98]Similar calculations were made for meat and milk.

BSN: for Cd, the model is based only on a human toxicological analysis[99], for a standard soil and for 4 soil destinations (including agricultural soil). Based on threshold values for tolerable daily oral/dietary Cd intake per kg bodyweight of 1 $\cdot$ 10^{-3} mg kg^{-1} (TDI), and for inhalatory Cd intake of 1.43 $\cdot$ 10^{-6} mg kg^{-1}, the maximum allowed amount of Cd in soil was then translated into a BSN.

These BSN were then compared with European and Belgian food- and fodder threshold values and adjusted if not stringent enough (but only for vegetables included in the food basket). In a next phase, these BSN were also checked against soil background values (of Cd) to ascertain that the difference between both would be detectable. Finally, in the Soil Decree, there is a formula to adjust BSN, BV, and TV for different soil types based on organic matter, clay content and pH (OVAM, 2004c).

For European food- and fodder threshold values for Cd, the Commission Regulation on foodstuffs (2006) based its values on an advice of 2 June 1995 where the Scientific Committee on Food (SCF) attaches its approval to a provisional tolerable weekly intake (PTWI) of Cd of 7 µg kg^{-1} body weight[100]. In the SCF report, the committee concluded that it was impossible to exclude carcinogenic risk from dietary exposure to Cd and was therefore unable to set a safe level of allowed Cd in foodstuffs. As a result, 7 µg kg^{-1} body weight was chosen as a very careful weekly level to keep dietary exposure to Cd as low as possible. In the meanwhile, more studies on the (non-) carcinogenic inhalatory and dietary uptake have become available, of which an overview is given in Bierkens *et al.* (2010).

---

[99]Cd is carcinogenic for inhalatory intake, and non-carcinogenic for oral intake. For non-carcinogenic contaminants, the Tolerable Daily Intake (TDI) concept is usually applied, which corresponds to the daily dose a person can be exposed to over a whole lifetime without harmful effects. For carcinogenic contaminants, the unit risk concept (excess cancer cases per unit of dose or unit of concentration) is used to derive an acceptable dose. Nevertheless, Cd is generally treated as non-carcinogenic (inhalatory and oral) because the threshold values for non-carcinogenic effects (kidney damage) are considered stringent enough to prevent carcinogenic effects (Bierkens *et al.*, 2010).

[100]PTWI is an endpoint used for food contaminants with cumulative properties, like Cd. The value of 0.007 mg kg^{-1} body weight represents allowed human weekly exposure to Cd unavoidably associated with the consumption of nutritious foods. This is identical to the TDI of 1 $\cdot$ 10^{-3} mg kg^{-1}.

The Codex Alimentarius[101] is in the Regulation on foodstuffs (2006) referred to as the basis for the establishment for values for Cd. The Codex (1995) mentions that: "*The maximum levels shall be set in such a way that the consumer is adequately protected". [] "It is desirable to have information about the contaminant concentrations in those foods or food groups that (together) are responsible for at least half and preferably 80% or more of the total dietary intake of the contaminant, both for consumers with average and high consumption patterns. Information about the presence of the contaminant in foods that are widely consumed (staple foods) is desirable in order to be able to make a satisfactory assessment of the contaminant intake and of risks associated with food trade. [] This problem, however, has to be addressed differently on a national and international scale. It is therefore important to have information about both average and high consumption patterns regarding a wide variety of foodstuffs, so that for every contaminant the most exposed consumer groups may be identified for every contaminant. Detailed information about high consumption patterns is desirable, both regarding group identification criteria (e.g. age or sex differences, vegetarian or regional dietary customs, etc.) and statistical aspects."* From this explanation, it is not clear on what percentage dietary intake of each of the crops maximum concentrations of Cd are based. The cautiousness is apparent through inclusion of the high consumption pattern through which the most exposed consumer group (*i.e.* the worst case scenario) is identified and taken into account.

The relation between soil contamination and health risk is represented as a curve with a positive slope. Humans are exposed to this health risk through

[101]The Codex Alimentarius Commission was created by FAO and WHO (1963) to develop food standards, guidelines and related texts such as codes of practice under the Joint FAO/WHO Food Standards Program. The main purposes are protecting health of consumers, ensuring fair trade in food trade, and promoting coordination of all food standards (governmental and non-governmental). The accession of the European Community should help strengthen consistency between standards, guidelines, and recommendations adopted under the Codex, and other relevant international obligations binding on the European Community and its Member States in the area of food standards (www.codexalimentarius.net;http://europa.eu/legislation_summaries/food_safety/international_dimension_enlargement).

consumption of crops grown on the land. Therefore, the steepness of the curve depends on the bioconcentration factor (BCF)[102] of the crop (crop 1, 2, and 3). The cut-off value for the risk assessment, *i.e.* the maximum allowed risk, is based on the precautionary principle (pp), providing adequate safety margins for human (or ecological) health. This cut-off value for maximum allowed risk is represented as a horizontal curve. The intersection of both curves results in the maximum allowed soil concentration ($C_{q1,\ 2,\ 3}$), so that, based on the BCF of the crop, the maximum allowed risk is not exceeded (Figure 4-9 (i)). Soil standards (C_{qs}) are then calculated from these food- and fodder threshold values, based on an average consumption basket of several crops (Figure 4-9 (ii)).

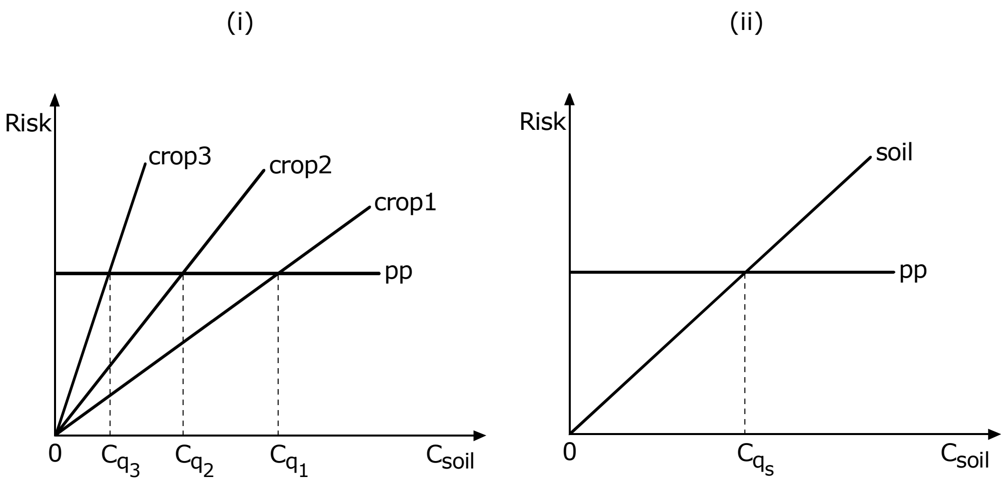


Figure 4-9 Theoretical representation of the determination of food- and fodder threshold values C_{q1}-C_{q3}(i), and soil standard C_{qs} (ii), represented as a maximum allowed soil concentration, based on the precautionary principle (pp)

In 4.1.5.4 we dig a little deeper into the fact that standards could reduce (soil) pollution in a cost-efficient and -effective way if they were chosen precisely at the level where marginal abatement cost (MAC) curves of remediation equal marginal damage cost (MDC) curves of contamination. However, such standards impose a high information- and control burden on authorities, very likely leading

[102]BCF = total element concentration in plant tissue ÷ total element concentration in the contaminated matrix (or the labile pool).

to a rather inconsistent follow-up. This is already obvious from the fact that instead of applying food threshold values for all land, authorities use a trapped system where a first control on soil standards determines whether food threshold values are checked.

In the phytoremediation model, we work with currnt food- and fodder threshold values and soil standards. Given that the application of soil standards is determined based on a correct average food basket (grown and consumed in the region), the use of soil standards should lead to the same results as the use of food- and fodder threshold values. This is analyzed in 0 where the model determines phytoremediation strategies, representing for each initial level of Cd contamination one remediating crop followed by one allowed food crop, based on soil standards. We also compare different soil standards: BSN, target value (TV), background value (BV), and the new BSN. In 4.1.5.6 we change and discuss the uncertainty level of exceeding current food- and fodder threshold values.

4.1.5.4 Alternative to the precautionary principle

The model based on current food- and fodder threshold values, assumes that these threshold values lead to economic efficiency. However, this might likely not be the case. We could step away from standards based on the precautionary principle (Figure 4-9) and instead base them on (i) conventional remediation (BATNEEC) technologies and (ii) plant-based technologies. Based on costs of both technologies, marginal abatement costs (MAC) could be determined. The marginal damage cost (MDC) of Cd in soil could be based on food intake and inhalation[103]. The economically efficient soil standard is then calculated as the intersection between the MAC curve of Cd in soil and the MDC curve of Cd in soil. Soil pollution should be abated up to the point where MAC = MDC. Any standard above this optimal soil standard results in additional damage costs of pollution which are higher than additional remediation costs. Any standard below

[103]Contaminated soil also results in damage of animals eating fodder grown on this land, resulting in metals accumulating in kidneys and liver. The value of this cost would then be equal to the loss in income by not being able to sell kidneys and liver, and destruction costs. We assume this is negligible.

this soil standard results in additional remediation costs above the additional damage cost.

Regarding the MDC, Nawrot *et al.* (2006) studied the relation between long-term changes in the internal Cd dose and simultaneously occurring mortality[104].

Table 4-29 Damage costs (TDC in € ha^{-1}) based on an economic valuation of elevated risk of lung cancer due to Cd exposure over 17.2 years (cancer incidence), assuming 8.2 disability adjusted life years (DALY) per lung cancer case, and a value of € 70,000 per DALY

Mg Cd kg^{-1} soil (C_x)	*10-10.6*		*5-10*		*1.5-5*		*0.9-1.5*	
	min	*max*	*min*	*max*	*min*	*max*	*min*	*max*
% Cancer	6	7.3	4	6	1.6	4	0	1.6
Cancer inc	0.19	0.23	0.12	0.19	0.05	0.12	0	0.05
DALY	1.53	1.86	1.02	1.53	0.41	1.02	0	0.41
TDC	106,764	129,896	71,176	106,764	28,470	71,176	0	28,470

Nawrot *et al.* (2006) have shown a significant association between risk of lung cancer and environmental exposure to Cd. Their findings suggest that continuing or past pollution from nonferrous smelters presents a serious health hazard. Table 4-29 presents the elevated lung cancer incidence per 100 people over 17.2 years based on the epidemiological study on inhalation of Cd. We then built further on these data, based on the number of (exposed) inhabitants with an average population in the municipalities Balen (273 inhab. km^{-2}, 72.9 km²), Lommel (307 inhab. km^{-2}, 102.4 km²), Overpelt (322 inhab. km^{-2}, 40.8 km²), and Neerpelt (371 inhab. km^{-2}, 42.8 km²). Assuming an equally spread population over different contamination levels, the number of inhabitants ha^{-1} is 3.1. The cancer incidence can then be calculated as the product of the elevated cancer incidence rate and the number of inhabitants per hectare. This results in DALY's, *i.e.* disability adjusted life years. According to Crettaz *et al.* (2002) one

[104]Yet another method to valorize the impact of contamination on soil value is the hedonic price method as used in Lewandowski *et al.* (2006), where the Willingness to pay (WTP) for clean land is determined. Clauw (2007) applied the hedonic price method to valorize the impact of contamination on private house prices. This indicates the WTP for clean private soil.

case of lung cancer results in 8.2 DALY's. Finally, the damage cost based on inhalation of elevated Cd concentration in soil can then be calculated based on a value of € 70,000 per DALY (Viscusi and Aldy, 2003; Van Wezel *et al.*, 2007). Applying linear interpolation we obtain a curve representing health costs of Cd inhalation (total damage costs, TDC). Based on these restricted data it seems that MDC is constant, *i.e.* TDC is a linear function and a raise in Cd concentration in soil results in a constant raise in TDC (based on inhalation). However, although MDC based on consumption of contaminated crops could not be valorized, these are assumed to rise (*i.e.* the damage caused by a rise in Cd concentration in soil rises). Therefore, we assume that the MDC has a positive slope (*i.e.* marginal damage costs rise when Cd concentration in soil rises).

These MDC are then compared with MAC. MAC = change in NPV(AGI) due to remediation (4%, over infinity). In case of conventional remediation, MAC is equal to the cost of conventional remediation (and the loss of one year income). In case of phytoremediation, MAC is the income loss over the years of phytoremediation. Costs of conventional remediation can be found in Table 1-2. At an average conventional remediation cost of € 25 per ton soil and 6,400 ton soil per ha, the cost of conventional remediation is € 160,000 per ha, regardless of the initial level of Cd contamination. This leads to the following two conclusions. Conventional remediation will not be an option when the cost of conventional remediation lies above health costs and once it is decided that conventional remediation is applicable, it does not matter anymore how far one remediates.

MAC of phytoremediation of Cd depends on the number of years of remediation, the income of the remediation crop, the alternative income on cleaned land and the interest rate (to calculate the NPV). Since we assume linear accumulation of Cd during phytoremediation, and a 100% effectiveness, the physical extraction of Cd is constant. This implies that at a given point in time, it takes as much time to go from 12 to 11 mg Cd kg^{-1} soil, as to go from 11 to 10 mg Cd kg^{-1} soil, resulting in the same income loss. This means that MAC of phytoremediation is constant for each C_0. Remediation then occurs for every C_0 where MDC > MAC and until MAC = MDC.

For example, if peas could have been grown on the soil (AGI' = € 4,010)[105], and it takes 11 years for willow (AGI' = € 110) to reach the next level of contamination (it doesn't matter whether we go from 11 to 10.8 mg Cd kg^{-1} soil or from 5 to 4.8 mg Cd kg^{-1} soil), then the NPV (4%) of the loss in income is € 42,000.

Since the income losses due to phytoremediation are potentially small, remediation could occur until very low levels of Cd concentration in soil have been reached. Phytoremediation will be chosen over conventional remediation when the net present value (NPV) of the loss in income is smaller than the cost of conventional remediation, which might be true in many cases. As a result, if phytoremediation would be supported as the best available technology, it would be economically efficient to continue remediation until Cd in soil reaches levels comparable to current soil standards.

An often heard comment is that when plant-based technologies are applied, contamination remains in soil for a longer time. Applying standards guarantees that only crops are grown so that no dangerous dose of Cd ends up in the food. Therefore, the fact that phytoremediation works slower than conventional remediation does not cause costs related to timing other than purely private ones (*i.e.* the fact that a high income crop can only be grown from a later time on) and this is taken care of in the decision model for this technology and its most suitable crop.

4.1.5.5 Comparison of soil standards and food threshold values

The base case phytoremediation model is based on food- and fodder threshold values. Here, the model is based on soil standards instead (Table 1-5).

[105]Mind that we do not mention a standard, because it is exactly this standard that we are determining in this exercise.

Table 4-30 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), soil standard (C_q = 1.2), *ceteris paribus*

C_0	C_q	*Rem. crop*	*Max* $R_{rem} + R_{HI}$	*r*
12-2.1	1.2 (endive)	EM	26,067	1,042.68
2.0-1.3	1.2 (endive)	W	30,064-118,297	1,202.56-4,731.88
1.1-0	1.2 (endive)	-	148,158	5,926.32

Current target values are 1.2 mg kg^{-1} soil. This means that if this condition is fulfilled, every crop from Table 4-1 should be allowed to be grown. This results in Table 4-30. Since soil standards are much lower than in the case of food- and fodder threshold values, remediation will at any case take very long. This is a disadvantage for willow, as this crop benefits from short remediation durations. As a result, energy maize is most often preferred as the alternative crop. In Figure 4-10 the predicted Cd concentrations in soils in the studied municipalities (Balen/Lommel, Overpelt, and Neerpelt) are shown (Clauw, 2007). Figure (ii) shows that the contamination is concentric around the smelters. It could therefore also be expected that farmers in the region are facing different concentrations of Cd, with farmers in the near vicinity of the smelters facing highest Cd concentrations.

(i) (ii)

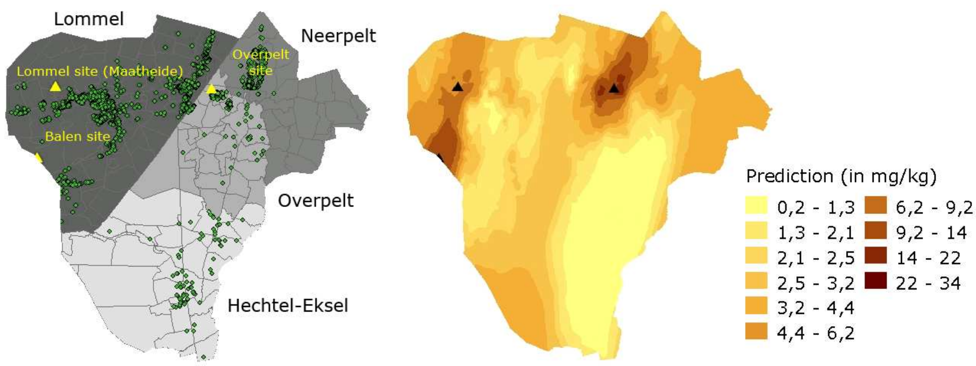


Figure 4-10 Prediction of Cd concentrations in soil (mg kg-1) in Balen/ Lommel, Overpelt and Neerpelt (ii), based on measurements (i) (Clauw, 2007)

It might therefore also be expected that the use of soil standards will benefit some farmers, whereas the use of food- and fodder threshold values will benefit others. This is confirmed in Figure 4-11 which demonstrates that the application of food- and fodder threshold values is beneficial for farmers when C_0 lies between 10.8 and 1.5 mg kg^{-1} soil.

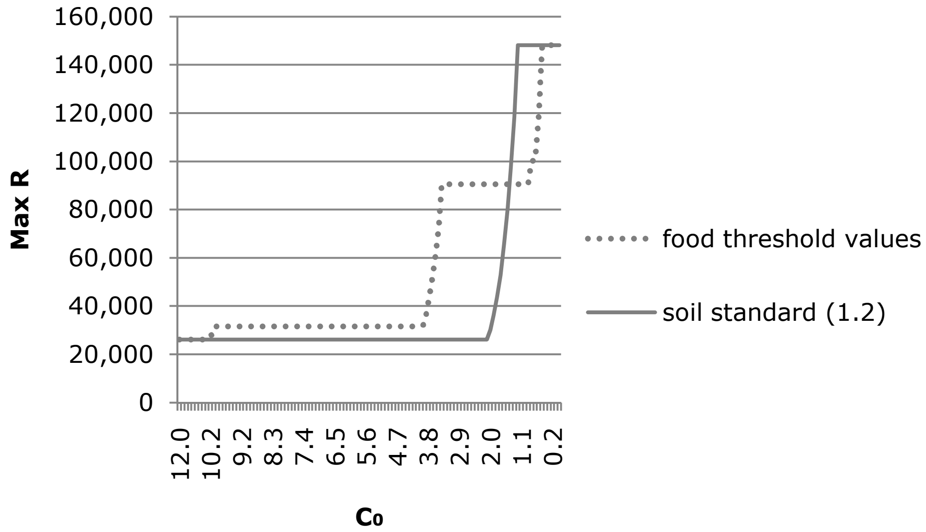


Figure 4-11 Comparison of maximum R over infinity (i=4%) for different initial Cd concentrations in soil (C_0), based on food- and fodder threshold values, and on TV (soil standard of 1.2), *ceteris paribus*, for each C_0

Soil standards and food- and fodder threshold values could lead to the same total welfare (depending on the distribution of agricultural land within the contaminated zone). This means that soil standards, which are based on an average food basket, are determined at the approximately right level. However, no conclusion could be reached about this since this is accidental, since the model’s decisions are not based on what is consumed but what is economically optimal to grow on the contaminated land.

Comparing different soil standards, we notice that this only results in a shift in R over the initial concentrations (C_0): the exact same R is reached at a higher C_0, in case soil standards are less strict (Figure 4-12).

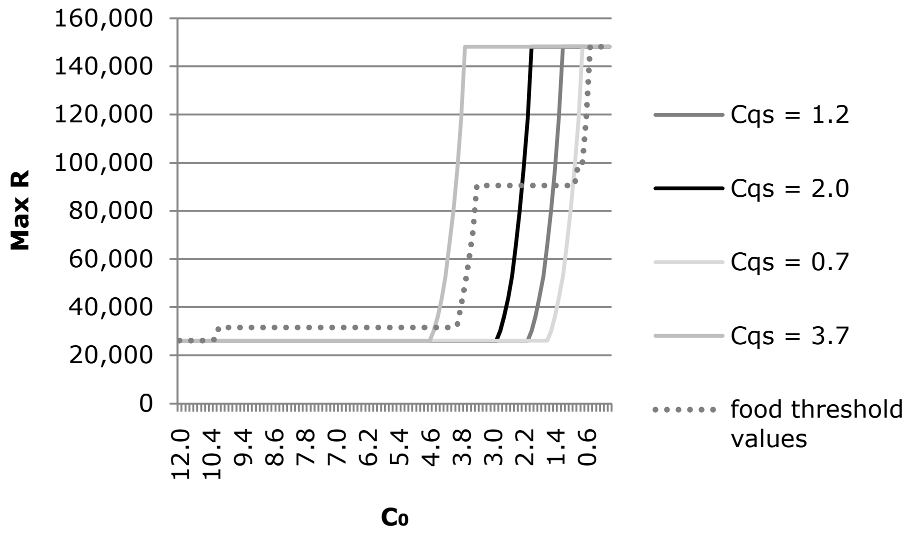


Figure 4-12 Comparison of maximum R over infinity (i=4%) for different initial Cd concentrations in soil (C_0), based on different soil standards (C_{qs}), *ceteris paribus*, starting from C_0

Figure 4-12 demonstrates that using soil standards or food- and fodder threshold values changes the pathway of optimal remediation and crop production and again that the total welfare reached by applying one or the other standard could be equal, given the correct circumstances related to the distribution of agricultural land in the contaminated region. The stricter the soil standard, the more contaminated land should be located in the less contaminated zone to lead to the same economic welfare.

We did not take into account control costs. As we might expect, these are higher for food threshold values than for soil standards. Hence we see in Flanders a current practice of applying soil standards before food threshold values. It seems as if soil standards only serve as a first criterion to determine whether food threshold values will be checked. An argument for the additional introduction of food threshold values after soil standards (and not just soil standards) is the fact that one could argue that from the moment crops are harvested, soil standards no longer apply. This would imply that soil standards should be applied first. When crops are harvested they should then comply with food- and fodder threshold values.

4.1.5.6 Uncertainty level

Much in the same way as for pH, Table 4-1 also shows maximum Cd concentration in soil for which the crop can be grown safely without exceeding Cd threshold values for food- and fodder use in 50%, 10% (base case), and 5% of measurements. The impact of a change in safety margin on maximum allowed soil concentrations differs for different HI crops. Shifting the pp curve up in Figure 4-9, we allow more risk[106], and threshold values per crop are higher. Shifting this curve down, we allow less risk, and threshold values per crop are lower. If pp in the base case ($pp_{10\%}$) is an overestimation, and $pp_{5\%}$ would be the correct amount of risk to be allowed, then we are being too stringent for the harvested crops, resulting in welfare loss, equal to the surface between the MDC curve for the food crop and the pp curves. If pp in the base case ($pp_{10\%}$) is an underestimation, and $pp_{50\%}$ would be the right amount of risk to be allowed, then we are being too tolerable for the harvested crops, also resulting in welfare loss.

We also analyzed whether this shift of pp, which has a different impact on C_q for the different crops since their MDC curves do not have the same slope (Figure 4-9), translates into a different economically most viable combination of remediation crop and HI crop.

Translating the 5% uncertainty level into maximum Cd levels in soil has an impact on the economically most viable option for very low levels of Cd concentration in soil (0.6-0.4 mg kg^{-1}), compared to the base case (Table 4-31).

[106] It is in fact not the pp curve that shifts, since the food- and fodder threshold values do not change. It is in the phase of converting these threshold values into maximum allowed soil concentrations that we allow more or less risk. However, moving the pp curve has the same effect and demonstrates the point we are trying to make.

Table 4-31 Results model R (€ ha^{-1}) and r (€ ha^{-1} y^{-1}), base case (uncertainty level of 10%) compared with uncertainty level of 5%, *ceteris paribus*

C_0	C_q	Rem. crop	Max. R_{rem} + R_{HI}	r
10% (base)				
0.6	0.5 (endive)	W	118,297	4,731.88
0.5-0.4	0.5 (endive)	-	148,158	5,926.32
5%				
0.6-0.5	0.8 (peas)	-	100,258	4,010.32
0.4	0.3 (endive)	W	118,297	4,731.88

If food- and fodder threshold values would apply consistently (*i.e.* apply for every crop, and not only when soil standards are exceeded), then the change in accepted uncertainty has an impact on the AGI adjusted for rotation and external benefits and costs (r) of farmers within the abovementioned zones (as compared to the base case), with reductions between € 722 and € 1,916 per hectare per year (Δr) (Table 4-32).

Table 4-32 Results model R (€ ha^{-1}) and the change in R (ΔR, € ha^{-1}) and r (Δr, € ha^{-1} y^{-1}) if the uncertainty level is reduced from 10% to 5%

C_0	R(10%)	R(5%)	Δ R	Δ r
0.6	118,297	100,258	(18,039)	(722)
0.5	148,158	100,258	(47,900)	(1,916)
0.4	148,158	118,297	(29,861)	(1,194)

We did not use the 50% uncertainty level in the model[107]. Cd background values in soil are 0.7 mg kg^{-1} in Belgium. Allowing an uncertainty level of 10% and 5% results in maximum allowed Cd concentrations for a lot of vegetables that lie below this background value. These concentrations seem to lead to an inconsistent separation of farmers in an unsafe versus safe zone. This should however lead us to critically analyze food- and fodder threshold values that lead to these soil standards, instead of allowing an uncertainty level of 50% to exceed food threshold values. Although using an uncertainty level of 50% would

[107]Although 50% was used for the crop advice in the Dutch Campine region (ABdK, 2008).

lead to higher maximum allowed soil standards (*i.e.* above background values), this would be a rather poor attempt of circumventing the problem of inconsistency between policies.

4.1.5.7 Compensation of farmers

Barde (2000) states that damage compensation is efficient when damage costs are correctly evaluated, when polluters and victims can be identified, when the causal relationship between pollution and damage can be established, and when the procedure of compensation does not entail excessive costs. However, if the number of victims is large, the efficiency criterion tells us that they should not be compensated. If people would be compensated for the external cost, there would be no incentive for them to not suffer from this external cost (Lofgren, 2000). Moreover, the Coase theorem states that the pathway to the optimal point of pollution depends on who has the property right. Does the neighborhood have the right on a clean environment and should they be compensated if the environment is polluted (Coase, 1960)?

In our case the damage has already been done, and the victims (we focus on farmers) were there before the pollution occurred and have not moved since then. Therefore, we suggest that farmers are compensated for their loss in income due to the corrective action they have to take because of the pollution. In 1997, the Flemish Government, Umicore, and the OVAM reached a common agreement (covenant) to deal with historical contamination caused by Umicore. In 2004, the same three actors signed an additional agreement in which it was stated that Umicore would invest € 39,000,000 in the remediation of the adjacent industrial and residential areas (total € 62,000,000), and that it would, together with the Flemish Government, contribute € 15,000,000 to deal with metals in the wider surroundings (total € 30,000,000) (De Turck, 2009).

The compensation for farmers is calculated as the difference between what they could have earned on clean soil and what they earn on the land which has to be decontaminated first with the alternative crop determined by the model, until the Cd concentration determined by the model is reached and the HI crop determined by the model can be grown. The model bases its decision on existing food- and fodder threshold values and then for each C_0 maximizes NPV(AGI) over infinity. The compensation per hectare per year is not just equal to AGI of the reference crop minus AGI of the remediating crop, because the HI crop after remediation suggested by the model is not necessarily the same as the reference crop. After remediation you might be able to grow a HI crop with a lower AGI than the reference crop (but was the best option suggested by the model since remediation until the Cd concentration where the reference crop can be grown would take too long) and this is taken into account in the calculation of the compensation.

Current agricultural activities in the four studied municipalities in the Campine region are summarized in Table 1-8. In the region, dairy cattle farming is the most important activity, with farmers growing fodder maize and temporary grass as feed for the winter period (61%). The other (main) activity is cereals (31%). These activities do not appear to be the economically most efficient ones, but rather find their incentive in farmers wanting to remain independent for feeding their cattle, convenience, uncertainty regarding price and demand of other crops, *etc.* (personal communication with farmers in the region, May 2009). Therefore, as a reference crop we should decide on a crop which would be grown if land were "clean", and if decisions were based on economic incentives.

We developed three cases based on which compensation for farmers is calculated: (i) phytoremediation based on food- and fodder threshold values, (ii) phytoremediation based on soil standards, and (iii) conventional remediation. In the first two cases, the model provides for each C_0 the most efficient NPV(AGI) by growing a remediation crop (from year 0 until year x), followed by a HI crop (from year x until infinity). The difference between case (i) and (ii) lies in the standards and thus the optimal pathway for each C_0. Both cases are compared with two references. In the first reference, NPV(AGI_1) represents what could have been earned if the economically most efficient crop allowed at a background level (BV) of Cd concentration in soil (0.7 mg Cd kg^{-1} soil) could have been grown from year 0 until infinity. In the second reference, NPV(AGI_2) could have been earned if the HI crop allowed at the economically efficient final Cd concentration in soil, determined by the model, could have been grown from year 0 until infinity. We include this second reference since it might just not be a possibility to compensate farmers because of what they might have grown if Cd concentration in soil would have been equal to background values. This second reference divides land into economically achievable end concentrations of Cd and bases the reference thereon.

For the third case (conventional remediation), NPV(AGI) is equal to the NPV(AGI) that could have been earned if the economically most efficient crop allowed at a background level of Cd concentration in soil could have been grown from year 0 until infinity minus the cost of conventional remediation. The reference for this third case is simply the case where the cost of conventional remediation would not have had to be made (and income on the land could have been generated one year earlier).

CASE 1: food- and fodder threshold value

Table 4-33 NPV(AGI) for different levels of C_0 (mg Cd kg^{-1} soil) as economically optimized by the model based on food- and fodder threshold values (C_q in mg Cd kg^{-1} soil), with NPV(AGI_1) when the economically most efficient HI crop (peas) allowed at BV (0.7 mg Cd kg^{-1} soil) could be grown immediately (Ref 1), and NPV(AGI_2) when the HI crop as determined by the model could be grown immediately (Ref 2) (4%, ∞, in € ha^{-1})

Contaminated site		*Ref 1*		*Ref 2*		
(1)	*(2)*	*(3)*	*(4)=(3)-(2)*	*(5)*	*(6)*	*(7)=(6)-(2)*
C_0	*NPV(AGI)*	*NPV(AGI_1)*	*ΔNPV(AGI_1)*	*C_q*	*NPV(AGI_2)*	*ΔNPV(AGI_2)*
12.0-10.1	26,067-26,611	100,250	74,183-73,639	10 (maize)	31,500	5,433-4,889
10.0-4.0	31,500	100,250	68,750	10 (maize)	31,500	0
3.9-3.5	32,586-71,505	100,250	67,664-28,745	3.4 (asparagus)	90,479	57,889-18,970
3.4-0.9	90,479	100,250	9,775	3.4 (asparagus)	90,479	0
0.8-0.7	100,250	100,250	0	0.8 (peas)	100,258	0
0.6	118,297	100,250	(18,047)	0.5 (endive)	148,158	29,853
0.5-0.0	148,158	100,250	(47,900)	0.5 (endive)	148,158	0

Table 4-33 provides an overview of the first case, where phytoremediation is used as a plant-based remediation technology until the economically most efficient food- and fodder threshold value is reached (column (5)). The resulting NPV(AGI) is shown in column (2), and is compared with column (3) and (6), with compensations equal to ΔNPV(AGI) (column (4) and (7)). Data of Table 4-33 are represented in Figure 4-13.

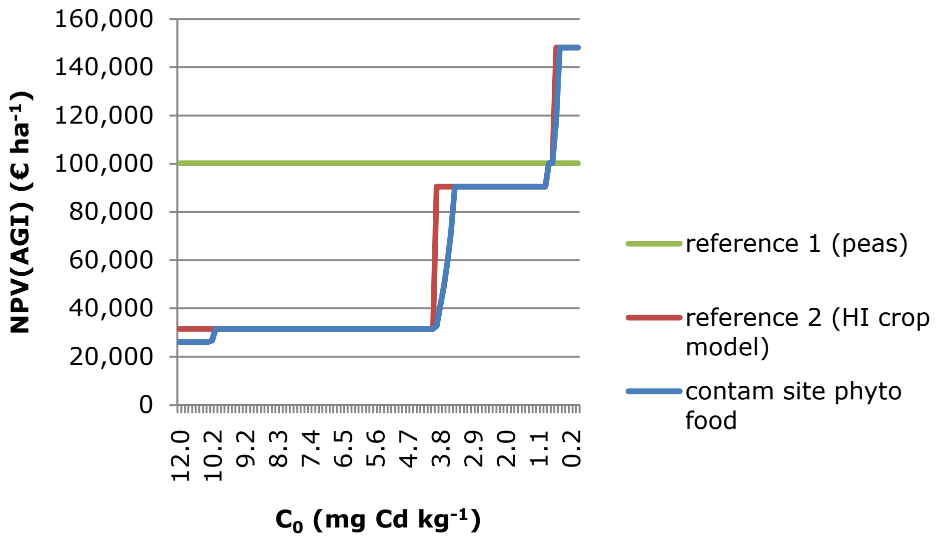


Figure 4-13 NPV(AGI) (4%, ∞, € ha^{-1}) (i) for different levels of C_0 as economically optimized by the model based on food- and fodder threshold values (contam site phyto food), (ii) when the economically most efficient HI crop allowed at BV could be grown immediately (reference 1), (iii) when the HI crop as determined by the model could be grown immediately (reference 2)

In Figure 4-13, the blue line indicates the economically most efficient NPV(AGI) for farmers for each initial level of Cd concentration (resulting from the model based on food- and fodder threshold values). The green line lies above this blue line until C_0 = BV, since from this Cd concentration peas can be grown immediately. The red line lies above the blue line between 12-10.1 mg kg^{-1}, between 3.9-3.5 mg kg^{-1}, and at 0.6 mg kg^{-1} and compares NPV(AGI) on contaminated land with NPV(AGI) when the HI crop decided by the model could have been grown immediately.

$$AF = \frac{i}{1-(1+i)^{-n}} \quad \text{(Eq. 31)}$$

The yearly compensation is calculated as the annuity that would lead to the difference in NPV(AGI) on contaminated and clean land. The formula for the annuity factor is given by **(Eq. 31**). This results in yearly compensations per hectare per year (Table 4-34). In case compensations are spread over the years when the remediation crop is grown, compensations are often very high. This approach will be preferred by current farmers. Spreading the compensations over infinity leads to the same final total compensation, but might be more acceptable by agents providing compensation.

Table 4-34 Compensation (€ ha^{-1} y^{-1}) per C_0 spread over the remediation period (comp rem) and spread over infinity (comp ∞), when Cd contaminated soil is remediated with plant-based technologies until food- and fodder threshold values have been reached in reference situation 1 and reference situation 2

Contaminated site	*Ref 1*			*Ref 2*		
C_0	*ΔNPV(AGI_1)*	*Comp rem*	*Comp ∞*	*ΔNPV(AGI_2)*	*Comp rem*	*Comp ∞*
12.0-10.1	74,183-73,639	2,967-3,273	2,967-2,946	5,433-4,889	217	217-196
10.0-4.0	68,750	2,750	2,750	0	0	0
3.9-3.5	67,664-28,745	4,173-6,092	2,707-1,150	57,889-18,970	3,570-4,020	2,316-759
3.4-0.9	9,775	391	391	0	0	0
0.8-0.7	0	0	0	0	0	0
0.6	(18,047)	0	0	29,853	6,326	1,194
0.5-0.0	(47,900)	0	0	0	0	0

Table 4-34 demonstrates the impact of the reference situation. The first reference is based on a unique reference concentration of Cd (*i.e.* a BV of 0.7 mg Cd kg^{-1}), and compensations will be due for all C_0 above this value. The difference between Comp rem and Comp ∞ lies in the duration of compensation payment. *E.g.* if C_0 is 3.5 mg Cd kg^{-1} soil, if the farmer follows the economically optimal pathway as suggested by the pathway, he will still lose a NPV(AGI) of € 28,745, compared to the situation where his land would have been clean.

Therefore, € 6,092 would have to be paid over 5 years, whereas € 1,150 would have to be paid over infinity. In the second reference, the reference concentration of Cd is adjusted based on what the model decided was economically best achievable. *E.g.* if C_0 is 0.6 mg Cd kg^{-1} there is a loss in NPV(AGI) equal to € 29,853 ha^{-1}, since the model decides that remediating with willow until 0.5 mg Cd kg^{-1} is economically most efficient. The reference is now based on NPV(AGI) that could have been earned if C_0 would have been 0.5 mg Cd kg^{-1}.

We continue with reference 1 since it is theoretically most correct and transparent (Figure 4-14). Farmers in a zone with Cd concentrations between 12 and 10.1 mg Cd kg^{-1} soil should receive a yearly compensation per hectare between € 2,967 and € 3,271 during the remediation period (> 59 years) if we assume peas could have been grown on clean land. The distribution of payments for different C_0 is represented in Figure 4-14.

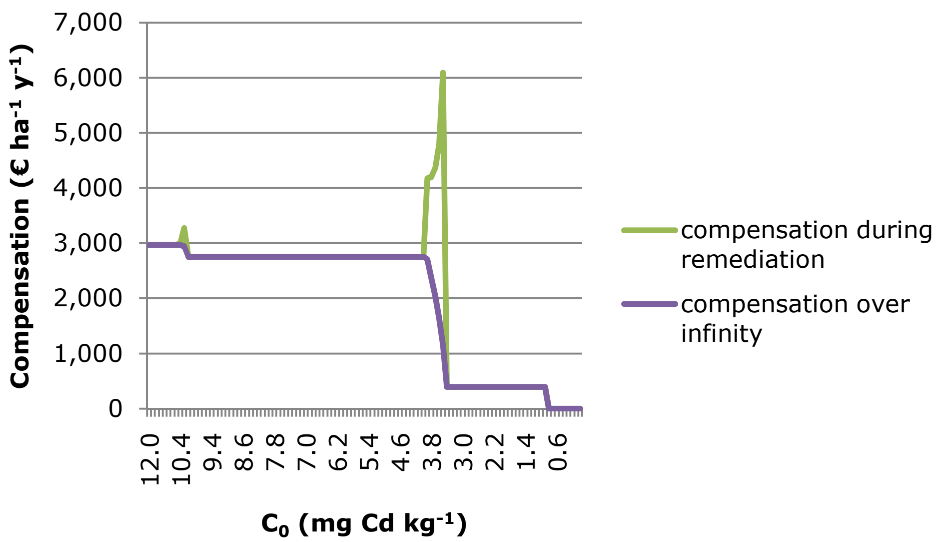


Figure 4-14 Compensation (€ ha^{-1} y^{-1}) per C_0 spread over the remediation period (comp rem) and spread over infinity (comp ∞), when Cd contaminated soil is remediated with plant-based technologies until food- and fodder threshold values have been reached, based on comparison with reference situation 1

Figure 4-15 zooms in on the peak in Figure 4-14. Farmers in a zone with C_0 between 3.9 and 3.5 mg kg^{-1} are encouraged by the model to grow willow to remediate their land until Cd concentrations in soil of 3.4 mg Cd kg^{-1} have been reached (to grow asparagus). Willow is suggested by the model because of the short distance to target (DTT). Farmers should receive high compensations during the duration of remediation because remediation periods are very short. Alternatively, spreading the compensation over infinity (which leads to the same total compensation) shows that these spread compensations follow the line of the previous compensations. Farmers in this zone should receive between € 4,173 and € 6,092 for 27 to 5 years (or between € 2,707 and 1,150 over infinity).

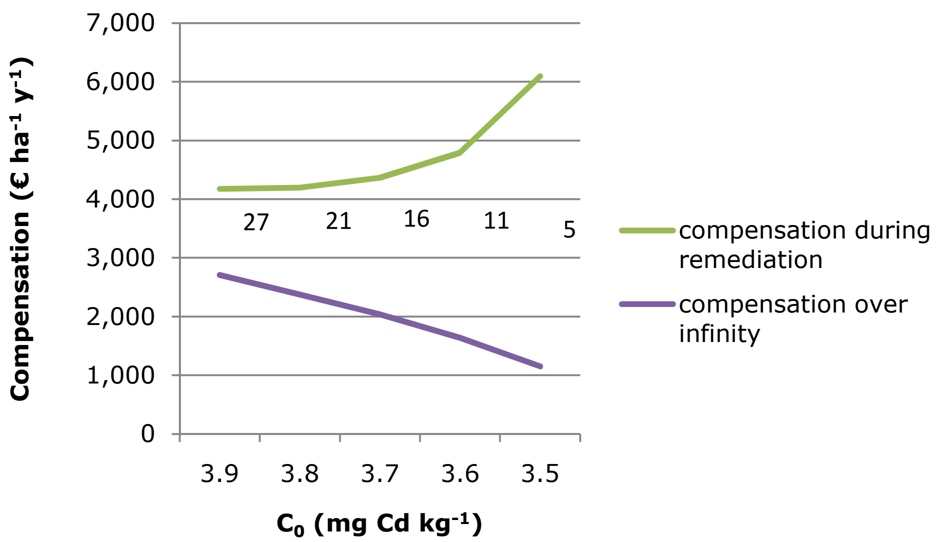


Figure 4-15 Compensation (€ ha^{-1} y^{-1}) for C_0 between 3.9 and 3.5 mg kg^{-1} spread over the remediation period (comp rem) with an indication of the time period, and spread over infinity (comp ∞), based on comparison with reference situation 1

CASE 2: Soil standard

Alternatively, the economically optimal crop choice could be based on soil standards. Since this results in different optimal pathways and NPV(AGI), compensations will differ compared to the situation where food- and threshold values are used. Figure 4-16 provides an overview of resulting NPV(AGI) in the second case, where phytoremediation is used as a plant-based remediation technology until the soil standard is reached. The soil standard of 2 mg Cd kg^{-1} is the standard from which remediation should occur, whereas the standard of 1.2 mg Cd kg^{-1} is the target value, this is the Cd concentration that should be aimed for on agricultural land. 3.7 mg Cd kg^{-1} represents the recently proposed new standard for Cd in soil. The green line represents NPV(AGI) of the reference situation (peas), allowed at the background value of 0.7 mg Cd kg^{-1}.

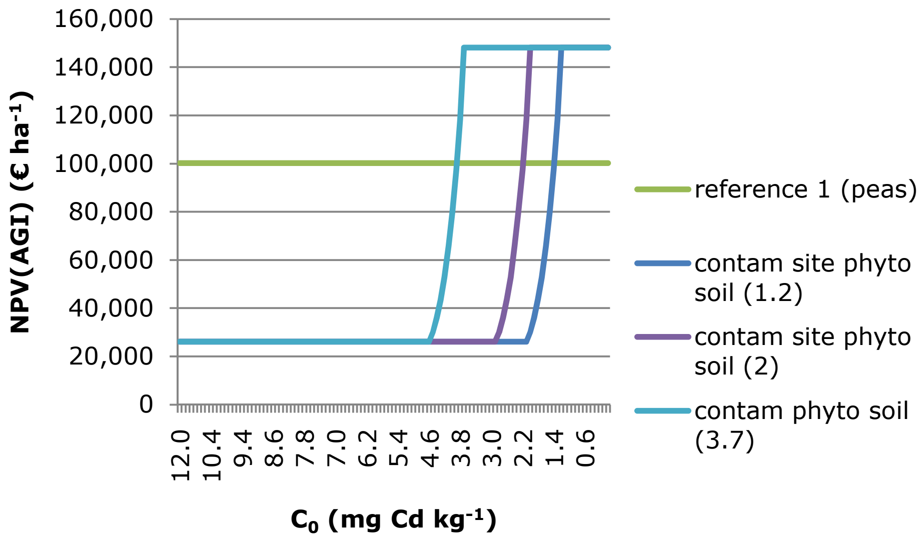


Figure 4-16 NPV(AGI) (4%, ∞, € ha-1) (i) for different levels of C0 as economically optimized by the model based on soil standards (1.2, 2, and 3.7 mg kg-1 soil), (ii) when the economically most efficient HI crop allowed at BV could be grown immediately (reference 1)

Resulting compensations for farmers are shown in Figure 4-17, with a peak a little before soil standards are reached. This is because willow is used for small DTT, which results in a higher NPV(AGI) than when another crop would have been grown, but results in the compensation being paid over a short period.

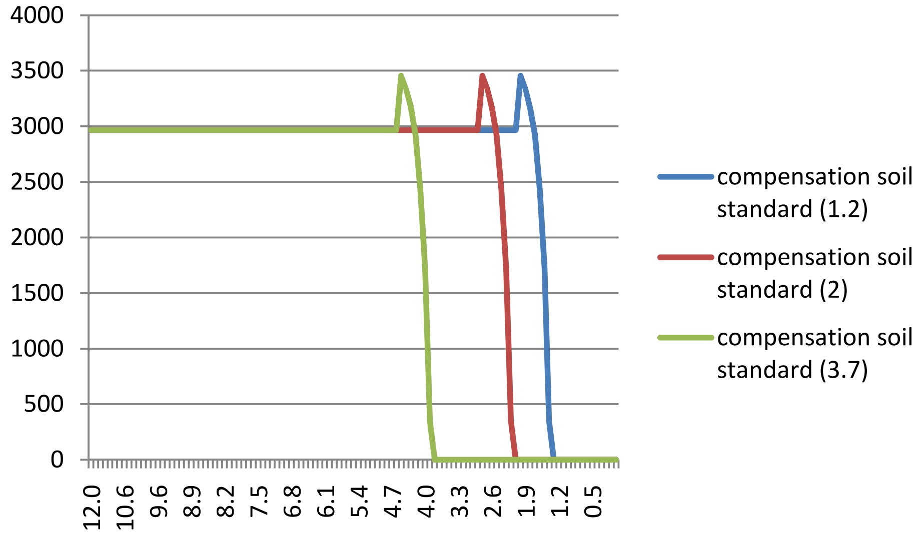


Figure 4-17 Compensation (€ ha^{-1} y^{-1}) per C_0 spread over the remediation period (comp rem) when Cd contaminated soil is remediated with plant-based technologies until soil standards (1.2, 2, and 3.7 mg kg^{-1} soil) have been reached, based on comparison with reference situation 1

CASE 3: Conventional

In the theoretical case of conventional remediation (which is not an option on agricultural land), the compensation would be a one-time payment of € 160,000 ha^{-1} or a compensation of € 6,400 ha^{-1} y^{-1} (over infinity).

Sensitivity analyses

The reference situation depends on the interpretation of "clean" soil. In the base case, the reference crop was fresh peas, since this is the economically most viable crop allowed at a Cd concentration (0.8 mg Cd kg^{-1} soil) above background value (0.7 mg Cd kg^{-1} soil). Stated otherwise, if BV are fulfilled, as we assume in the reference situation, it would be economically optimal to grow peas. In Table 4-35, oignon/cabbage is added as a reference crop, since these are the economically optimal crops if in the reference situation the target value (1.2 mg Cd kg^{-1} soil) would be fulfilled. Endive is added as the reference crop if we assume that in the reference case Cd concentrations in soil are that low that the overall economically most optimal crop could have been grown, which is endive (0.5 mg Cd kg^{-1} soil).

Table 4-35 Compensation (€ ha^{-1} y^{-1}) per C_0 spread over the remediation period (comp rem), comparing NPV(AGI) (between brackets) when Cd contaminated soil is remediated with plant-based technologies until the economically most efficient food- and fodder threshold values have been reached with NPV(AGI) of different reference crops

C_0 contaminated site	*Peas (100,250)*	*Oignon/cabbage (68,850)*	*Endive (148,150)*
12.0-10.1 (26,067-26,611)	2,967-3,273	1,711-1,877	4,883-5,402
10.0-4.0 (31,500)	2,750	1,494	4,666
3.9-3.5 (32,586-71,505)	4,173-6,092	2,236-0	7,127-16,243
3.4-0.9 (90,479)	391	0	2,307
0.8-0.7 (100,250)	0	0	1,916
0.6 (118,297)	0	0	6,326
0.5-0.0 (148,158)	0	0	0

If we consistently overestimate AGI of all crops (remediation- HI crops, and reference crops), the effect on compensations is straightforward: we will overestimate compensation with the same percentage.

In Figure 4-18 the ratio of compensations based on soil standards and food- and fodder threshold values over the remediation duration is shown. Differences in compensations between the different standards can be explained because the model suggests other HI crops after remediation, which do not coincide with the reference crop. In general, the compensation when soil standards are used lies 8% higher. For a soil standard of 3.7 mg Cd kg^{-1}, compensations go down from 4.0 mg Cd kg^{-1}, whereas they start rising when food- and fodder threshold values are used. For soil standards of 2 and 1.2 mg Cd kg^{-1} the ratio peaks a little before these standards are reached.

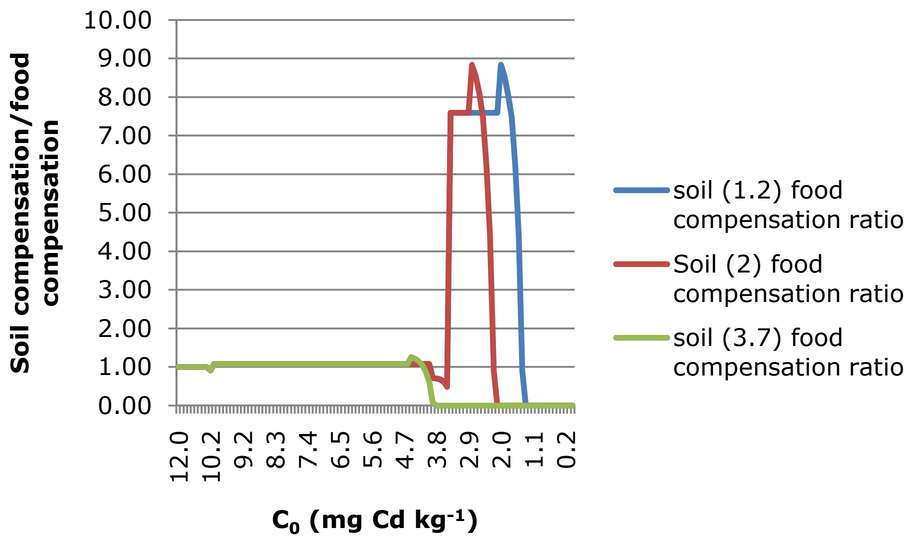


Figure 4-18 Ratio of compensations per C_0 when the model is based on soil standards (1.2, 2, and 3.7) and when the model is based on food- and fodder threshold values

We cannot draw any conclusions on which standard (soil or food) will lead to the lowest total compensation for the whole Campine region since we do not know the correct distribution of Cd concentrations over the region. Moreover, conclusions on whether to use food versus soil standards should not be based on a minimization of compensations, but on a maximization of NPV(AGI) for each C_0 (remember that this is not the same since the optimal HI crop for each C_0 could be different from the reference crop). Another important consideration in this context is that all calculations are based on AGI, of which the relation with the agricultural income is explained in Chapter 3.1.

Chapter 4.2 *Energy maize as an acceptable alternative crop for risk management of contaminated land*

This chapter has been published in:
Thewys, T., Witters, N., Ruttens, A., Van Slycken, S., Meers, E., Tack, F.M.G., Vangronsveld, J. (2010) Economic viability of phytoremediation of an agricultural area using maize: impact on the farmer's income. *Int. J. Phytorem.*, 12, 7, p. 650-662

Thewys, T., Witters, N., Meers, E., Vangronsveld, J. (2010) Economic viability of phytoremediation of an agricultural area using maize: economics of anaerobic digestion of heavy metal contaminated maize in Belgium. *Int. J. Phytorem.*, 12, 7, p. 663-679

Parts of this chapter have been submitted in:
Van Slycken, S., Witters, N., Meers, E., Peene, A., Michels E., Adriaensen, K., Ruttens, A., Vangronsveld, J., Du Laing, G., Thewys, T., Tack, F.M.G. (xxxx) Safe use of metal contaminated agricultural land by cultivation of energy maize (*Zea mays*). Environ. Pollut.

Abstract

This chapter elaborates the specific case of plant-based technologies with energy maize in an agricultural area where most farmers raise dairy cattle and grow fodder maize. The first part of this chapter deals with the economic viability of using energy maize as an alternative crop and as a replacement of fodder maize as an income generating crop. The acceptance of plant-based technologies for remediation is, besides the extraction rate, determined by the effect it has on the income of the farmer whose land is contaminated. This income can be supported by producing renewable energy through anaerobic digestion of energy maize, a crop that extracts only relatively low amounts of metals, but that can be valorized as a feedstock for energy production. The effect on the income per hectare of growing energy maize instead of fodder maize seems positive, given the most likely values of variables and while keeping the basic income stable,

originating from dairy cattle farming activities. We suggest growing energy maize to aim at risk reduction, to generate an alternative income for farmers, and in the very long run also to realize a gradual reduction of the pollution levels. In this way, remediation is demoted to a secondary objective with sustainable risk reducing land use as primary objective. The second part of this chapter explores the economic opportunities for the farmer of digesting the harvested contaminated biomass himself, by performing a Net Present Value (NPV) analysis on the digestion activity and by calculating the probability of a positive NPV of income resulting from the digestion installation. We investigate the trade-off between the maximum price for energy maize that can be paid by the digestion activity and the minimum price that the farming activity needs to compensate for covering its production costs. Integrating the first part in the second part results in an increase in total extra income for the farmer (*i.e.*, from both growing energy maize and performing digestion).

4.2.1 Introduction

In this chapter, we investigate the economic viability of energy maize. Energy maize (*Zea mays*) is a crop with low metal uptake capacities (Meers *et al.*, 2005b; Zhang and Banks, 2006), but it has the advantage to produce a high biomass (leading to a moderate absolute extraction), and the local agricultural sector is very familiar with growing this crop. Energy maize is used for energy production purposes rather than for conventional applications as food or feed. In Europe, production of energy maize, used for biogas production through anaerobic digestion, is increasing rapidly. Specific *Zea mays* cultivars are being selected/bred for optimal biogas production potential much in the same way as conventional cultivars were selected for their nutritive characteristics. As such, energy maize and biogas production represent a relatively new branch of agriculture, which has emerged at a large scale over the last five to ten years.

The farmers involved in this case study are mainly dairy cattle farmers who wish to continue their activities. In the Campine, an average farmer possesses 40 ha of land. Twenty ha are used for fodder maize, 20 ha are used as grassland for cattle (based on FOD Economy: Statistics, 2006). The economic model presented in this chapter consists of two scenarios. The effect of the switch from fodder to energy maize on the dairy cattle activities is described in the first part of this chapter. In this first scenario, the cost of buying clean fodder maize has to be compensated by revenues coming from selling energy maize to a digester. We calculate the "change in income" (Z) for the farmer by switching from fodder maize to energy maize and the probability that the farmer's income will at least be sustained by the switch (Prob(Z > 0)). The energy maize can be used as a feedstock in a dry digestion installation, *i.e.,* an installation fed with biomass with a high (>15%) dry matter (dm) percentage. In the second part of this chapter we analyzed whether it is economically achievable for local farmers to run a digestion installation fed with metal contaminated biomass. As such, the farmer has two main activities: he grows energy maize while continuing to raise cattle (that are fed with unpolluted fodder maize bought from outside the contaminated area), but he is also managing a digester (in cooperation with other farmers). Extra revenue might then be generated for the farmer by the digestion activity. The total change in income (Z) can then be calculated as the sum of the extra income from switching from fodder to energy maize and the extra income from the energy production.

4.2.2 Economics of switching from fodder to energy maize

4.2.2.1 Data and methods

Maize and metals

Maize has optimal growing conditions on the sandy soils in the Campine at a pH between 5.0 and 6.0 (De Boer *et al.*, 2003). The biomass production of energy maize on the trial field in the Campine is 20 ton dm ha^{-1}. At a dry matter percentage of 28–33%, this translates into a fresh matter yield of 60 ton fresh matter (fm) ha^{-1} (Table 3-1). Combined with the acceptance of the farmers (maize being a conventional crop) and its economic opportunities for non-food applications, there is an incentive to investigate this crop in more detail.

Economics of energy maize used as an alternative crop

The farmer grows energy maize instead of fodder maize while continuing his dairy cattle farming activities and marketing milk products at the same level as before. Therefore, he needs the same amount of fodder maize as before. The farmers involved are mostly dairy cattle farmers who desire to continue their current activities. On average, they have 40 ha acreage, consisting of 20 ha grass (no risk) and 20 ha fodder maize to feed dairy cattle. Their basic income comes mainly from milk (FOD Economy: Statistics, 2006). The 20 ha that were used for fodder maize are now used to grow energy maize which is sold for energy production purposes. To feed cattle, uncontaminated fodder maize is bought from outside the contaminated area. By growing energy maize instead and buying fodder maize from outside the contaminated area, the basic income of the farmer will be diminished with purchase and transport costs of fodder maize and supplemented with revenues originating from the higher yield of energy maize and converting the latter into energy through anaerobic digestion. The cost per hectare of growing energy maize is the same as the cost of growing fodder maize (personal communication with farmers in the region, Aegten, and external firms, 2009). These conditions are necessary for the assumption that the reference basic income per hectare of the farmer originating from dairy cattle rearing (€1,123 ha^{-1}) does not change. The reference basic income will then be supplemented or reduced with the alternative risk reducing activities.

Table 4-36 Comparison of the current economic situation with the economic situation when growing energy maize

Situation before energy maize	*Situation during energy maize*
Fodder maize *(BP_{FM})* at cost *C* per ha	Energy maize *(BP_{EM})* at cost *C* per ha
Selling milk products at price *M*	Selling milk products at price *M*
	Buying fodder maize *(BP_{FM})* at price *P*
	Selling energy maize *(BP_{EM})* at price *P*
	Transport cost of energy and fodder maize per ton per kilometer *(T_d)*
	Support for energy crops *(S)*[108]

[108]This has been abolished.

The model variables are represented in Table 4-36 and show that reclamation of the soil is economically viable only if revenues from selling contaminated energy maize exceed the cost of buying clean fodder maize. This depends on the relative yield of fodder and energy maize, their prices and the transport costs. Farmers receive a price between € 1,800 and 2,000 ha^{-1} for fodder maize, depending on the yield, or a price per ton fm (P) of € 30 ton^{-1} in the base case. The production cost of maize is approximately € 1,200–1,250 ha^{-1} (De Boer *et al.*, 2003). At a most likely fm yield of respectively 50 and 60 ton ha^{-1}, fodder and energy maize will therefore not be sold below € 24–25 ton^{-1} fm and € 20–21 ton^{-1} fm respectively. We assume that energy maize will be sold immediately, implicating that no extra ensiling is necessary. Transport costs are studied separately as total transport costs differ for energy and fodder maize. Respective transport distances D_1 and D_2 are calculated in (Eq. 35) and (Eq. 36) in appendix A to this chapter. Given the transport cost per ton per kilometer (T), the total transport costs are then calculated as follows (i) for energy maize: $T_{EM} = D_1 \cdot T_d \cdot BP_{EM}$; and (ii) for fodder maize: $T_{FM} = D_2 \cdot T_d \cdot BP_{FM}$. Until recently, due to the transition of fodder maize to energy maize, the farmer received an extra support (S) for growing energy crops of maximum € 45 ha^{-1} from the Agency for Agriculture and Fisheries (ALV). As a result, the extra revenue (Z) from soil management activities per ha energy maize, including the compensation for transport costs, is given by the following formula:

$$Z = P \cdot (BP_{EM}-BP_{FM}) + S + (D_1 \cdot T_d \cdot BP_{EM} \cdot A - D_2 \cdot T_d \cdot BP_{FM} \cdot A)/A$$
$$Z = (P+D_1 \cdot T_d) \cdot BP_{EM} - (P+D_2 \cdot T_d) \cdot BP_{FM} + S \qquad \text{(Eq. 32)}$$

With:

P = price of fodder and of energy maize per ton fm (€ ton^{-1} fm)

BP_{EM} and BP_{FM} = yield of energy and fodder maize respectively, per ha (ton fm ha^{-1})

S = the (abolished) energy premium per ha (€ 45 ha^{-1})

T_d = transport cost per ton per kilometer (€ ton^{-1} km^{-1})

A = total number of ha remediated (number of participating farmers $\cdot$ 20 ha)

4.2.2.2 Results

Deterministic approach

If the extra revenue, Z, is positive, the income of the farmer is raised due to the risk reducing soil management activities. When Z is negative, the income goes down due to these activities. We assume the following numerical values for the determining variables: A =300 ha (and accordingly D_1 = 1.38 km and D_2 = 8.8 km), BP_{EM} = 60 ton ha^{-1}, BP_{FM} = 50 ton ha^{-1}, T_d = € 0.5 ton^{-1} km^{-1}, P = € 30 ton^{-1} and S = € 45 ha^{-1}. According to (Eq. 32), we then come to an extra income for the farmer of Z = € 166.5 per ha remediated. This result is conditioned by the implicit assumption that all determining variables are measured with full certainty, reflected in using only one numerical value for each of them. However, in reality, all these variables have numerical values belonging to a range, *i.e.*, we have to take account of uncertainty in calculating the extra revenue Z. Therefore, Monte Carlo simulations using the model are performed.

Taking into account uncertainty

To take into account uncertainty about the numerical value of determining variables we use the technique of Monte Carlo simulation (using the software package Crystal Ball, Decisioneering Inc.). A run in this simulation calculates Z according to values randomly taken from the presupposed value ranges for predefined variables. Performing numerous runs (in our study 20,000), the Monte Carlo technique calculates numerous values for the net extra revenue Z, based on probability distributions for the determining variables defined by a minimum, maximum and most likely value (Table 4-37). The distributions of the variables have a triangular shape. Such a distribution is usually employed when there is insufficient data to fit any other distribution but when minimum, maximum, and most likely values are known or presupposed based on expert information. The value for a specific determining variable is then obtained as a randomly drawn value from this distribution. The most likely value is the value of the deterministic approach. Minimum and maximum values form a range of ±10% starting from the most likely value. The yield of energy maize however does not follow this distribution. In the base case, energy maize has a minimum yield, which is the same as the yield of fodder maize, the most likely value is 20% better, and the maximum value is 30% better. We assume that fodder maize has a certain yield of 50 ton fm ha^{-1}. Moreover, as opposed to the deterministic approach, prices for energy and fodder maize are no longer equal in the Monte Carlo simulations: prices are correlated (correlation coefficient of +0.5). In 50% of the cases run during the simulations, the prices of energy and fodder maize move in the same direction (average positive linear relation).

Table 4-37 Base Case values of the determining variables and forecast result for the extra income of the farmer, all with a ± 10% range, accounting for uncertainty

Determining Variable	*Min*	*Most likely*	*Max*
A (number of ha occupied by energy maize)	270	300	330
T_d (transport cost in € ton^{-1} km^{-1})	0.45	0.5	0.55
P (price of fodder maize in € ton^{-1} fm)	27	30	33
P (price of energy maize in € ton^{-1} fm)	27	30	33
BP_{EM}/BP_{FM} (relative yield of energy to fodder maize)	1	1.2	1.3
Forecast	*Min*	*Most likely*	*Max*
Extra income farmer (€ ha^{-1} remediated)	-303.4	113.8	505.5

Given these assumptions, indicated as the "base case", the average extra revenue per ha is € 113.8. Compared to the revenue before the risk reducing land management (€ 1,123 ha^{-1}), this is an increase of more than 10%. The probability that the average extra revenue per ha is not negative, is 82.6%, which indicates the probability that the farmer's income will not decrease. Monte Carlo sensitivity analysis shows the relative importance of the different variables in explaining the variance of the extra income (Table 4-38). The first variable, the relative yield of energy to fodder maize, accounts for approximately 81.3% of the variance in forecast values of the extra income Z. A large yield of energy maize per hectare compared to fodder maize has a large positive impact on Z. It is obvious that the price of energy maize has a similar–although smaller–positive effect. If the price of fodder maize rises, the expenditure for externally buying increases. The price of energy maize is more important than the price of fodder maize as energy maize involves more ton ha^{-1}.

Table 4-38 Explanation of the variance of the extra income (Z)

Variable	*Contribution to the variance of Z*
BP_{EM}/BP_{FM} (energy to fodder maize yield)	81.3%
P (price of energy maize)	13.4%
P (price of fodder maize)	-4.9%

Sensitivity analysis

Next, the effect of significantly changing the most likely value—and its surrounding range—on the amount of the farmer's extra income and on the probability of obtaining a positive extra income is investigated. The sensitivity of the extra income by changing the most likely values of the yield of energy maize in ton fm per hectare (BP_{EM}) (Table 4-39), and the price per ton fm (P) of energy and fodder maize (Table 4-40) is calculated. In Table 4-41, the assumption that fodder and energy maize have an equal price or are correlated, is discarded. This means that prices of fodder and energy maize can change independently of each other. The minimum price of energy maize is then calculated, (i) given the condition that the probability of obtaining a positive extra income coming from this alternative activity is at least 90%, and (ii) given the values of the base case for the price of fodder maize and the other variables (Table 4-37). Calculations in Table 4-41 are motivated by the fact that the energy maize is contaminated. This might have a negative effect on its price.

Sensitivity farmer's income to the relative yield of energy and fodder maize

In the pessimistic scenario, the relative yield of BP_{EM} to BP_{FM} is changed in a negative way compared to the base case. Keeping BP_{FM} at the value of the base case (50 ton ha^{-1}), the extra revenue Z goes down to € 12.5 ha^{-1}, a reduction with 89% compared to the base case. In the optimistic scenario, the minimum yield of energy maize is at least 10% higher than fodder maize. This has a positive impact on the extra revenue per hectare, which is now € 166.5 ha^{-1}, a raise of 46% compared to the base case. Growing energy maize has a reasonable probability to be economically viable when the most likely value for the yield of energy maize per hectare is 20% higher than the yield of fodder maize, accompanied by a maximum yield of energy maize that is 30% higher.

Table 4-39 Average extra income per ha (Z in € ha^{-1} y^{-1}) and probability of a positive extra income (Prob(Z>0)), given changes in the relative yield of energy and fodder maize (BP_{EM}/BP_{FM}), *ceteris paribus*

Scenario	*Variable*	*Min*	*Most likely*	*Max*	*Average Z*	*Prob(Z>0)*
Pessimistic	BP_{EM}/BP_{FM}	1	1.1	1.2	12.5	55.0%
Base case	BP_{EM}/BP_{FM}	1	1.2	1.3	113.8	82.6%
Optimistic	BP_{EM}/BP_{FM}	1.1	1.2	1.3	166.5	96.5%

Sensitivity of the farmer’s income to the price of energy and fodder maize (P)

In Table 4-40, the price of maize per ton is equal for energy and fodder maize. In the base case Z = € 113.8 ha^{-1} and Prob(Z > 0) = 82.6%. The minimum price is set at € 24 ton^{-1} fm, which is equal to the production cost per ton fodder maize. As can be seen, a change in price does not have a large effect on the probability of positive extra revenue coming from the risk reducing soil management. When the farmer grows energy maize, he has a fair chance (75–87%) to sustain and even increase his income, depending on the height of the prices (for now assumed equal for energy and fodder maize), within the assumed range of € 24 to € 36 ton^{-1} fm. Comparing Table 4-39 and Table 4-40, it is confirmed that Z is much more sensitive to changes in the relative yield BP_{EM}/BP_{FM} than to changes in the price of maize. When P increases, revenue increases, this however is largely neutralized by the larger expenses for fodder maize. The same conclusions (but to a lesser extent) can be obtained by comparing the effect of changes in the relative yield BP_{EM}/BP_{FM} and in the price of maize on Prob(Z > 0). The probability that the extra revenue is positive, *i.e.*, that the farmer’s income does not diminish by the phytoextraction activity, is more sensitive to a change in relative yield than to a change in the price of the maize. Changes in BP_{EM}/BP_{FM} lead to high uncertainty (*i.e.*, low probability on a positive income): it can generate high revenues accompanied by a high certainty level, but it can also lead to very low revenues with a low certainty level. Price changes, on the contrary, keep the income more stable and do not lead to a much lower probability of a positive income.

Table 4-40 Average extra income per ha (Z in € ha^{-1} y^{-1}) and probability of a positive extra income (Prob(Z>0)), given changes in the price of energy and fodder maize - assumed equal (P in € ton^{-1} FM), *ceteris paribus*

P(maize) (€ ton^{-1})	*Average Z (€ ha^{-1})*	*Prob(Z>0)*
24 ±10%	65.8	75.2%
27 ±10%	90.2	79.8%
30 ± 10%	113.8	82.6%
33 ± 10%	140.9	85.9%
36 ± 10%	165.3	87.3%

Determining P(energy maize), given P(fodder maize) such that Prob(Z > 0)≈ 90%

The assumption that prices of fodder and energy maize are correlated is now discarded and the minimum price of energy maize is determined, given a determined price of fodder maize and the values for the other variables as in the base case (Table 4-37). The prices calculated for energy maize have to assure that the probability of a positive extra income (Prob(Z > 0)) coming from growing energy maize is very high (≈90%). The yield of fodder maize is kept fixed at 50 ton ha^{-1}. The yield of energy maize (relative to the yield of fodder maize) is indicated by the different lines in Figure 4-19 (Optimistic, Base Case, and Pessimistic scenario, see Table 4-39). The price of fodder maize is indicated on the X-axis. By considering several values for these variables, we can calculate the price of energy maize that renders the probability to obtain Z > 0 ≈ 90%. This means that the income of the farmer per hectare from energy maize would very probably remain at least at a status quo relative to the income before soil management. Table 4-41 shows for the base case BP_{EM}/BP_{FM} how the middle curve on Figure 4-19 is determined.

Table 4-41 Calculating the energy maize price and average Z, given BP_{FM} =50, BP_{EM}/BP_{FM} as in the base case, for different P(fodder maize), to have a Prob(Z>0) ≈ 90%

P (fodder maize) *(€ ton^{-1}, given)*	*P (energy maize)* *(€ ton^{-1}, calculated)*	*Average Z* *(€ ha^{-1}, calculated)*
24 ± 10%	25.5 ± 10%	151.2
30 ± 10%	31.2 ± 10%	183.9
36 ± 10%	37 ± 10%	223.6

In Table 4-41, the price range of energy maize is calculated given a price range of fodder maize, to have at least a 90% probability of a positive Z and with BP_{EM}/BP_{FM} = 1.2 as in the base case. This is done for three price ranges of fodder maize, resulting in three price ranges for energy maize and three different Z's (each with a probability of 90%). In Figure 4-19, each curve represents a different relative yield of energy to fodder maize, resulting in different levels of the necessary prices for energy maize to reach a probability of 90% that the farmer's income will not decrease.

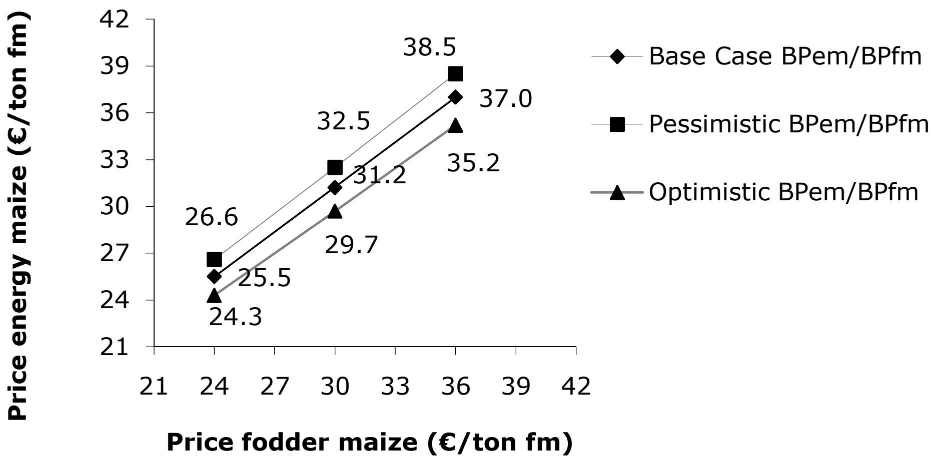


Figure 4-19 Determination of energy maize price, given a fodder maize yield of 50 ton fm, to have an almost certain positive extra income (Prob(Z>0) ≈ 90%), and differing according to the relative yield of energy maize to fodder maize (BP_{EM}/BP_{FM}) and P(fodder maize)

From Figure 4-19, it is clear that the necessary price of energy maize depends on its yield relative to the yield of fodder maize (fixed at 50 ton fm ha^{-1}). In the pessimistic scenario, the difference between energy and fodder maize prices must be at least € 2.5 ton^{-1} fm (38.5–36) to obtain an almost certain positive extra revenue *Z*. In the base case scenario, this minimum difference is lowered to € 1 ton^{-1} fm (37–36). In the optimistic scenario, the price of energy maize can even be € 0.8 ton^{-1} fm lower (35.2–36). Moreover, if the overall level of prices is higher, this obviously increases Z. If the farmer does not receive these calculated prices (due to contamination), the probability of a positive Z will be lower than 90%, so chances are increasing that the alternative activity might reduce the income of the farmer. The figure should be read as follows. If, given a base case relative yield BP_{EM}/BP_{FM} (*i.e.,* min. 1; most likely 1.1; max. 1.2), the farmer has to buy fodder maize at a price of € 30 or € 36 ton^{-1} he wants to receive a price of at least € 31.2 or € 37 ton^{-1} respectively. Given the base case relative yield, the relation between fodder maize price and energy maize price can be expressed as follows:

$$P(EM)= 0.96 \cdot P(FM) + 2.48 \qquad \text{(Eq. 33)}$$

4.2.3 Economics of anaerobic digestion of energy maize

4.2.3.1 Data and methods

Anaerobic digestion of contaminated energy maize

Anaerobic digestion is the conversion process where biomass is converted by bacteria into methane in four phases in the absence of oxygen. The end products of the digestion process are biogas and digestate. Due to its high energy content, the resulting biogas can be utilized in engines and machines that work on natural gas, be used as a transport fuel, or even be injected in the natural gas distribution network (Verstraete, 1981; Ramage and Scurlock, 1996). We opted for the first choice, burning the gas in a gas engine with heat recovery, a Combined Heat and Power engine (CHP). The other product that comes out of the digester is the digestate, a mixture of water and stabilized organic matter. All metals present in the biomass end up in this digestate. There is no such thing as "the" digestate. In general, digestate is a good alternative for chemical fertilizer and in addition, it is more stable than undigested manure, with a better humus performance (Timmerman *et al.*, 2005). Information, data, and studies relating to the potential influence of metal contents in the biomass on the digestion process are scarce, and have been described in Chapter 3.2. The residual digestate after digestion is further processed by separation into a liquid fraction (2 dm%) and a solid sludge fraction (25 dm%). The solid fraction is subsequently dried (85 dm%) using the heat recuperated from the CHP unit which is powered by the biogas generated by the anaerobic process. The fate of metals during post-processing of the digestate and their manipulation has been described in Chapter 3.3.

Economic model

To reduce risks associated with cultivating fodder crops on diffusely metal enriched agricultural land, the farmer will switch from fodder to energy maize while continuing his dairy cattle raising activities at the same level and while continuing marketing dairy products as before. The basic income of the farmer is supplemented or reduced with the positive or negative income from growing energy maize and energy activities (anaerobic digestion). The economic viability is evaluated by calculating the extra positive or negative income stemming from the digestion activity. The Net Present Value (NPV) of the stream of yearly net incomes (the yearly cash flow (CF)) is calculated over the lifetime of the digestion installation. A project is accepted when the present value of the net income stream over its lifetime is positive. To calculate NPV, information is needed on investment costs, yearly expenses, and yearly revenues, based on several variables. The time scale (t) is 20 years. For the digestion activity, a discount factor (i) of 6% is assumed (Maeng *et al.*, 1999; Murphy and McKeogh, 2006). (Eq. 34) gives the formula for the NPV. CF_0 is the initial investment cost of the project. CF_t is the cash flow in year t (t: 1, …, n). From this NPV the yearly extra income is calculated (appendix B of this chapter).

$$NPV = CF_0 + \sum_{t=1}^{n} CF_t/(1+i)^t \qquad \text{(Eq. 34)}$$

To continue business as usual for his dairy activities, the farmer requires the same amount of fodder maize as before. A fresh matter (fm) yield of 50 and 60 ton per hectare respectively for fodder and energy maize is assumed. Fodder maize will have to be bought from outside the contaminated area. The cost of growing maize is independent of whether it is fodder or energy maize (€ ha^{-1}). The basic income (from growing maize and selling milk products) will therefore be the same as before and is altered by buying fodder maize and selling energy maize. The economic viability of reclamation of the soil depends on the yield of fodder and energy maize, their relative prices and transport costs. Extra revenue might be generated by the digestion activity.

Variable description

Number of farmers: dimension of digester and CHP engine ((Eq. 38) and (Eq. 41))

The optimal number of participating farmers can be derived from the investment cost and thus dimension of the engine. The investment cost in the CHP is a logarithmic function of the size of the engine, where economies of scale apply starting from an engine with an electric capacity of 900 kWe ((Eq. 42) and (Eq. 43)) (appendix C of this chapter). Given the base case values for the variables, an engine with a capacity of 900 kWe results in an optimal number of participating farmers of 13.6. For ease of calculation, we use 15 as the most likely value in the base case.

Price and relative yield of energy maize

From the point of view of the digesting activity, energy maize is the feedstock. Costs involve the price of energy maize, the transport cost ton^{-1} km^{-1} (T_d) and the ensiling cost. Transport costs (appendix A of this chapter) are on behalf of the buyer of the biomass and as such are a revenue for the vendor of the biomass. The cost of ensiling varies between € 55 and € 93 ha^{-1} according to De Boer *et al.* (2003) and the Animal Sciences Group (2006). Consistent with these estimates, the study performed by Goossens (2007) for OWS (Organic Waste Systems) assumes that the ensiling cost amounts to € 2 ton^{-1} fm. Currently, farmers receive a price between € 1,800 and 2,000 ha^{-1} for fodder maize. We assume a price for energy maize of € 1,800 ha^{-1}, a yield of 60 ton fm per ha, and a price of € 30 ton^{-1} fm. The distribution of the yield of energy maize is determined relative to the yield of fodder maize, *i.e.,* min. 1 (= 50/50), most likely 1.1 (= 55/50) and max. 1.2 (= 60/50 ton ha^{-1}). The price and the relative (*i.e.,* compared to fodder maize) yield of energy maize can be changed in the NPV-model.

Digestate

The digestate contains metals—resulting from the uptake performed by maize—so a solution has to be sought with respect to the proper disposal or processing. In Chapter 3.3 we combusted the digestate to replace for cokes, after the maximum amount of digestate had been used for fertilizing purposes. In our calculations, this was the economically most viable option. In this chapter, we opt for the currently most applied processing of the digestate, *i.e.*, separation followed by drying and proper disposal. Personal communication with Fillip Velghe of Biogas-e, a non-profit organization for the promotion of digestion in Flanders, teaches us that the separation cost of the digestate is included in the initial investment cost of the digester at € 10 per ton input. Operating costs for separation vary around € 2 per ton digestate. The drying cost of the digestate consists for a major part of energy costs. This heat does not have to be bought, because it suffices to use all net produced heat. This means however that no heat can be sold. An extra drying cost of € 10 per ton solid fraction is used in calculations. Transport costs of the dried solid fraction are about € 3 per ton. The disposal cost of the dried solid fraction is estimated at € 5 ton^{-1} (the cost of disposal of the contaminated digestate can however become negative, indicating an income from selling the digestate).

Yearly revenues

In Flanders, every electricity supplier is obliged to deliver a specific amount of electricity generated from renewable sources. The producers of green electricity receive a green current/electricity (GC) certificate for every MWh (net) produced. Concerning green electricity produced by digestion, this refers to the electricity available net of the use in the digestion process, as indicated by a decision of the Flemish Government in 2004 (Energy Decision, 2004) and clarified by the Flemish Regulation Entity for the Electricity and Gas Market (VREG, 2007). The producers receive a minimum guaranteed price of € 80 per certificate, guaranteed during 20 years. The current market price is situated at approximately € 112.5 per MWh, with only slight deviation from this number over recent years.

Another official incentive policy involves support for exploiting a gas engine in a CHP system. This system promotes that, besides the electricity produced, heat will be recovered, for which the government issues CHP Certificates during 10 years. The minimum guaranteed price per certificate is € 27. Again, the market price is higher than the guarantee and is currently situated at approximately € 40.5 per MWh, with only slight deviation from this amount over recent years (appendix D of this chapter).

It is assumed that all net heat produced (Eq. 39) (*i.e.*, after only 4.1% is used by the digester since it concerns dry digestion and large investments in insulation are made by this specific installation) will be used. In the base case, 100% of the net heat will be used to dry the digestate. However, if less heat is necessary to dry the digestate, the surplus heat can be sold and thus has an opportunity value. More specifically, the use of natural gas in a boiler can be omitted. This means a reduction in cost of € 27.5 MWh^{-1} heat, depending on the gas price.

Net electricity (Eq. 40) can be used locally or can be put on the grid. In the first option, the opportunity value of electricity is obtained by multiplying the price normally paid for electricity by the farmer (most likely value of € 100 MWh^{-1} in the base case), with the sum of the amount normally used and the volume that can be sold at local consumers of electricity. The second option results in revenues from selling net electricity produced (*i.e.,* after process use) to the grid. It is assumed that all net electricity produced is put on the grid. In the base case, the selling price of electricity to the grid is € 80 MWh^{-1}, being 80% of day-ahead electricity trading prices.

4.2.3.2 Results

Extra income using deterministic approach

In the base case, the yearly CF from year 1 onwards, necessary to calculate the NPV are as indicated in Table 4-42. Percentages are calculated relative to the total incoming and outgoing CF. The initial investment costs in year 0 are not shown explicitly. This base case results in a mean NPV of € 266,271. Per year and per hectare occupied by energy maize, this means an average extra income stemming from the digesting activity of € 77.4 ha^{-1}, *i.e.*, the income that the digestion project can pay to the farmer for it still to be accepted as an economically viable project. This is an extra income, additional to the revenue from growing and selling energy maize (€ 166.5 ha^{-1} in the deterministic approach).

Table 4-42 CF in year 1 and NPV for the digester, given most likely deterministic values for the variables

		Absolute value (€)	*%*
(1)	*Total CF in*	*1,624,108*	*100%*
	Electricity sold to the grid	523,190	32%
	Opportunity value electricity	0	
	GC Certificates	735,735	45%
	Opportunity value heat	0	
	CHP certificates	365,184	23%
	Other support	0	
	Digestate	0	
(2)	*Total CF out*	*1,319,422*	*81%*
	Capital cost digester	233,305	14%
	Capital cost CHP engine	109,823	7%
	Maintenance CHP	143,289	9%
	Maintenance digester	38,700	2%
	Feedstock (energy maize), incl. transport	552,438	34%
	Ensiling energy maize	36,000	2%
	Digestate cost	97,687	6%
	Diverse costs	108,180	7%
(1)-(2)	*CF in - CF out*	*304,686*	*19%*
(3)	*NPV over 20 years (discount rate of 6%)*	*266,272*	

<u>Extra income taking into account uncertainty</u>

To take into account uncertainty about the numerical value of determining variables we use the technique of Monte Carlo simulation (using the software package Crystal Ball, Decisioneering Inc.). A run in this simulation calculates the NPV according to values randomly taken from the presupposed value ranges for predefined variables. The value ranges are defined as the most likely value ± 10%. Performing numerous runs (in our study 20,000), this technique calculates numerous NPV's of the net results, resulting in a distribution of the NPV's together with the probability to obtain a positive NPV (Prob(NPV>0)). An analysis of this NPV indicates the most important variables determining profitability. Next, the most likely values of the most determining variables are changed in a negative way and their ranges accordingly (*ceteris paribus*) and the simulation for the NPV is run again. The results indicate the sensitivity of the NPV of the extra income from the digestion activity with respect to significant changes in the numerical values of the determining variables. Moreover, these variables are used to calculate the maximum price for energy maize as a feedstock that can be paid by the digestion activity not to affect profitability. The variable specific minima, maxima, and most likely values used in this study are found in Table 4-43. Given the assumptions in Table 4-43 (called the "base case"), the average extra income per ha is € 76.6, to be compared with € 77.4, the result calculated in the deterministic case. The probability that the average extra income per hectare is not negative is 75.7%. Sensitivity analysis shows which variables contribute the most to the uncertainty of the forecasted average extra income.

Table 4-43 Base Case value ranges for the variables determined by a most likely value ± 10%

Variables	*Minimum value*	*Most likely value*	*Maximum value*
(1)	(2)	(3)	(4)
Number of farmers (N)	13.5	15	16.5
Yield energy maize (BP_{EM}) (ton ha^{-1})	50	60	65
Price energy maize (P) (€ ton^{-1})	27	30	33
Price energy maize per ha (€ ha^{-1})	1,350	1,800	2,145
Transport cost maize (€ ton^{-1} ha^{-1})	0.45	0.5	0.55
Price CHP Certificates (€ MWh^{-1})	36.45	40.5	45
Price GC Certificates (€ MWh^{-1})	101.25	112.5	125
Opportunity value heat (€ MWh^{-1})	24.75	27.5	30.25
Price electricity grid (€ MWh^{-1})	72	80	88
Disposal cost digestate (€ ton^{-1})	4.5	5	5.5

Table 4-44 shows that the variability in the price of GC certificates accounts for approximately 39% of the variance in forecast values and can be considered the most important determining variable in the model. As can be seen in the same table, the price of GC certificates, the price of energy maize, and the price of electricity sold to the grid together account for 87% of the variance in Z. In the next section, the effect on the average extra income per hectare (Z) and on the probability of getting a positive extra income from the digester of changing the most likely values of these three variables is calculated (*ceteris paribus*).

Table 4-44 Determination of important variables in calculating the average extra income per hectare per year (Z)

Variable	*Contribution to the variance of Z*
Price GC Certificates	38.8%
P(energy maize)	-24.9%
Price electricity sold to the grid	23.6%

<u>Sensitivity analysis</u>

<u>Less favorable values for the determining variables</u>

In the pessimistic scenario, the most likely values (and accordingly the minimum and maximum values) of the different income determining variables are changed in a negative way (*ceteris paribus*) compared to the base case. In doing this, we maintain the same distribution as in the base case (most likely value ± 10%). In Table 4-45, column (4) we see the average extra revenue which can be earned by the digestion activity per hectare per year. In column (5) we see the result for the probability of obtaining a positive NPV and thus a positive Z.

Table 4-45 Effect of a change in the most likely value of a variable (col. 2- 3) (*ceteris paribus)* on average Z (col.4) and on Prob(NPV>0) (col.5)

		Most likely		*Z*	*Prob(NPV>0)*
	(1)	(2)	(3)	(4)	(5)
	Variables	*Base case*	*Worst case*	*Value (€ $ha^{-1}y^{-1}$)*	*Value (%)*
(1)	Price GC Certif. (€ MWh^{-1})	112.5	101.25	-79.3	22.2%
(2)	Price energy maize (€ ton^{-1})	30	33	-63.2	28.0%
(3)	P electricity grid (€ MWh^{-1})	80	72	-49.2	31.6%
(4)	Price CHP Certif. (€ MWh^{-1})	40.5	30	-38.2	35.6%
(5)	Number of farmers	15	10	-24.8	40.7%

Row 1 of Table 4-45 confirms the importance of the GC Certificates for the economic viability of a digestion installation. If more certificates are traded on the market, prices might decrease with 10% and render the installation economically unfeasible. Moreover, these numbers confirm that the minimum price (€ 80 MWh^{-1}) for GC Certificates *(ceteris paribus)* will certainly (in this project) not be able to render the installation viable. The price of energy maize is equally important. The large impact of a 10% raise in energy maize price on Z and on Prob(NPV > 0) is shown in row 2. In row 3, the price of electricity is lowered to a level where it was in 2008. The price of electricity on the day-ahead market is far from constant. The effect of a 10% lower price reduces Z with approximately € 126 (from € 76.6 in the base case to € -79.3). The most likely value in the pessimistic scenario of the price of CHP Certificates (row 4) is set at € 30 MWh^{-1}, with a minimum of € 27 MWh^{-1}. This minimum guarantee leads to a probability of a positive NPV of 35.6%. Compared to the GC Certificates, a guaranteed minimum price for CHP certificates does provide a larger probability of a positive extra income. If the most likely number of farmers is reduced from 15 (base case) to 10 (row 5), the probability of a positive NPV of the digestion activity is reduced from 75.7% (base case result) to 40.7%. This can be explained by the fact that investment costs in digester and CHP engine are too high to be compensated by the yearly net CF generated by a lower number of farmers. This is due to not taking advantage of economies of scale occurring in dimensioning the CHP engine.

Digestate

The contaminated digestate will first be separated and then dried. The economic aspects concerning drying the digestate are complicated. In the base case, it is assumed that all heat is used to dry the digestate. As a consequence, no net heat produced is sold while at the same time CHP certificates are granted for all net heat produced. In practice however, a digester does not receive certificates for the heat used to dry that part of the digestate coming from energy crops (if we assume that crops coming grown on contaminated land are considered energy crops). Therefore, in the base case all net heat is actually sold (*e.g.,* to be used by a nearby swimming pool), the same amount of heat is bought at the same cost to dry the digestate (thereby costs and revenues resulting from the heat flow are compensating). As a result, certificates are granted for all net heat, as it is not used to dry the digestate but for external application, *i.e.,* the swimming pool.

In this part of the analysis, we will examine what happens if no certificates are granted for the heat used for drying the digestate. This heat thus generates no extra revenue, and no certificates are granted for it. The rest of the net heat is sold at a price of € 27.5 MWh^{-1}, and certificates are granted for this part.

Table 4-46 Effect of using heat for drying digestate on Z

% heat				*Z*	*Prob(NPV>0)*
digestate	*process*	*sold*	*certificates*	*€ ha^{-1} y^{-1}*	*%*
25%	4.1%	71.9%	71.9%	271.4	99.5
38%	4.1%	57.9%	57.9%	143.7	90.7
50%	4.1%	45.9%	45.9%	23.6	58.0
75%	4.1%	20.9%	20.9%	-232	1.7

Table 4-46 shows that when only 25% of net heat is needed to dry the digestate and 72% can be sold, there is a 99% chance to have a positive extra income from the digester. If, however, 75% of net heat produced is needed to dry the digestate, extra revenues for the farmer are negative. If this is the case, a construction as in the base case might prove helpful, *i.e.,* find local demand for all net heat, sell it, and buy the amount of heat at the same price needed to dry the digestate. In doing this, farmers will receive certificates of € 40.5 MWh^{-1} for all net heat. In the dry digestion process, 38% of the heat can be used to dry the digestate and still render the installation very probably profitable (see row 2).

In the base case, it is assumed that the disposal cost of the contaminated digestate is € 5 ton^{-1} digestate. Additional drying costs are € 10 ton^{-1} solid fraction. Additional calculations show that this drying cost has to go down to € 6, *ceteris paribus,* in order to have a (Prob(NPV) > 0) ≈90%, resulting in an extra income of € 130.4 per hectare per year. Disposal costs can be € 2.2 ton^{-1} digestate for the installation to have an almost certain positive effect on the income of the farmer (Prob(NPV) > 0) ≈90%, resulting in an extra income of € 136.2 per hectare per year. From the moment that the digestate can be disposed off at no cost or even be sold, there is a 98% chance on a positive extra income with an average extra income of € 213.3 per hectare per year.

Maximum price for energy maize as a feedstock as to Prob(NPV>0)≈90%

What is the maximum price for energy maize that can be paid by the digester such that the probability of a positive NPV of the results from digestion lies above the 90% range? To constrain conditions, the value of the NPV is calculated using the least favourable values for the variables.

Table 4-47 Maximum energy maize price (col. 3) and average yearly revenue per ha (col. 5) such that Prob(NPV>0) ≈ 90% given the base case (row 1) and negative scenarios (rows 2-4)

	Variable	*Most likely (given)*	*Max Price EM (calc)*	*Prob(NPV>0) (%)*	*Z (€ ha^{-1} y^{-1})*
	(1)	(2)	(3)	(4)	(5)
(1)	Base Case	Table 4-43	28.7	90%	134.6
(2)	Price GC Certif. (€ MWh^{-1})	101.25	25.5	90%	123.5
(3)	Price CHP Certif. (€ MWh^{-1})	30	26.1	90%	127.6
(4)	Price electr. grid (€ MWh^{-1})	72	26.1	90%	128.2

In column (2) of Table 4-47, we find the most likely pessimistic values for the different chosen variables. In column (3), the maximum price that the digester can pay for energy maize per ton fm to have an almost certain positive NPV, is given. This Prob(NPV > 0) appears in column (4). In column (5), the average revenue originating from the digestion process expressed per hectare of energy maize is calculated. If for example, prices for GC Certificates are lowered with 10%, the maximum price of energy maize that can be paid by the digester is € 25.5 ton^{-1}. This low price is caused by the fact that we want a 90% guarantee that the farmer's income is sustained, resulting in an average extra income of € 123 ha^{-1} year^{-1}.

There is a conflict of interest in the determination of the price of energy maize. On the one hand, farmers want to receive a price for their energy maize that is as high as possible. On the other hand, the price of energy maize is an important cost element of the digestion activity, which is in the hands of the same farmers. The exact price (calculated by Prob(Z > 0) ≈90%) from which the energy maize producer will sell to a digester–depending on BP_{EM}, BP_{FM}, P(fodder maize) and T_d—is represented in Figure 4-19. The calculated prices that the digester can pay are compared with the price that the farmer wants to receive as a producer of energy maize.

Putting the prices for energy maize found in Table 4-47 (representing maximum prices for energy maize for a digester to have a 90% chance to be profitable) in (Eq. 33) (representing the relation between fodder and energy maize price for farmer to have a 90% chance to sustain his income) gives maximum prices of fodder maize in € ha^{-1} between 24.0 and 27.3. Within this range, energy maize will be sold by the farmer to the digester at prices determined in Table 4-47. Consequently, the farmer selling his energy maize to the digester has a 90% chance to increase his income and at the same time, the digester exploited by the farmer has a 90% chance to be profitable. However, remember that production costs of maize are € 1,200 ha^{-1}, resulting in a cost of € 24 ton^{-1} fodder maize, a price below which fodder maize will not be sold.

In the next part, it is shown however that the condition of a Prob(Z > 0) ≈90% simultaneously in both scenario's is too stringent in the integrated scenario. In the integrated scenario where the farmer produces energy maize, sells it to its own digester, and produces electricity, heat, and digestate, the probability of a positive extra income coming from both activities is cumulated.

Integration: total income from cultivating and digesting energy maize

Collecting all influences from the first and second scenario, the technique of Monte Carlo simulation allows the model to calculate the simultaneous influence on the income of the farmer of all considered determining variables. Table 4-48 (rows 1–4) gives an overview of the contribution of the determining variables to the variance of the forecasted extra income of the farmer (columns 2–4). The columns refer to the extra income resulting from selling the energy maize (column 2), the income from the digestion activity (column 3), and the total extra income for the farmer combining the two activities (column 4).

Table 4-48 Integration of scenario 1 and 2: Contribution of the variability of the variables to the variance of the total extra revenue of the farmer per ha per year (Z)

	Variables	*income selling energy maize*	*income digestion*	*total extra income*
	(1)	(2)	(3)	(4)
(1)	Yield energy maize	81.3%		52.3%
(2)	Price energy maize	13.4%	-24.9%	
(3)	Price fodder maize	-4.9%	-6.5%	-12.2%
(4)	Price GC Certif.		38.8%	19.5%
(5)	Price electricity to grid		23.6%	11.4%
(6)	Z (€ ha^{-1} y^{-1})	113.8	76.6	191.4
(7)	Prob(Z>0)	82.6%	75.7%	89.0%

Given the base case values in the first scenario (Table 4-37), there is an 83% chance for an improvement of the farmer's income, resulting from growing and selling energy maize (row 7, column 2). The average extra revenue for the farmer then varies around € 114 ha^{-1} $year^{-1}$ (row 6, column 2). Given the base case values from the second scenario (Table 4-43), there is a 75.7% chance of obtaining a positive income from the digestion activity for the farmer (row 6, column 3), which means an average extra income per hectare of about € 76.6 ha^{-1} (row 6, column 3). Integrating both parts, Monte Carlo simulation results in a total average extra revenue for the farmer (*i.e.,* from both growing energy maize and also performing digestion) of € 191.4 ha^{-1} $year^{-1}$ (row 6, column 4). Moreover, the probability that this total extra income is positive is almost 90% (row 7, column 4) when the farmer grows energy maize instead of fodder maize and simultaneously exploits the digester (in cooperation with other farmers). These results are based on the base case values of energy and fodder maize prices of € 30 ton^{-1}. Such an extra income would increase the actual labour income per hectare per year originating from dairy cattle rearing (approximately € 1,123 ha^{-1} $year^{-1}$ in 2005) by more than 15%.

Taking a closer look at the determining variables reveals that the variability of the yield of energy maize per hectare contributes for 52.3% to the variance of the total forecasted extra revenue (row 1, column 4). Therefore, current research is ongoing for selecting *Zea mays* cultivars based on their optimal biomass and biogas production potential. As such, energy maize and biogas production represent a new branch of agriculture. Prices of fodder and energy maize have a correlation of 0.5 in the integrated model, resulting in the relative large importance (12%) of the price of fodder maize. For the integrated model to reach a 90% probability that the extra revenue is positive, it suffices that both prices have a most likely value of € 30 ton^{-1}. The price of GC certificates is an important variable in explaining the variance in the income resulting from the digester (column 3, row 4). In the integrated model, it is still clear that subsidies will continue to have a large impact on the viability of the sustainable land use and management project. Prices of electricity are very volatile, given the rather important variance of the income explained by this variable (11%), contracts with electricity distributors might offer a stronger guarantee to maintain economic viability.

4.2.4 Conclusion

In the case of vast areas, although only moderately polluted, the economic aspect indicates the opportunity for the low cost plant-based technologies. In choosing energy maize one opts for a long-term scenario. It is a choice for sustainable land use and management instead of remediation as such

(Vangronsveld *et al.*, 2009; Meers *et al.*, 2010). Energy maize is not an obvious alternative crop within the context of soil remediation. The choice is justified by the fact that it is (i) a known crop with a (ii) high energy production potential.

The first part of this chapter calculates the extra income (Z) of the farmer when energy maize is grown instead of fodder maize. Moreover, it calculates the probability that the income of the farmer will at least be sustained after the switch (Prob(Z > 0)). Results from this chapter reveal a promising outlook for the farmers in the Campine region as it appears that the average extra income during risk reducing land management is positive, *i.e.*, the income is at least sustained. In 2005 farmers have an average income per hectare fodder maize of € 1,123 ha^{-1}. Given deterministic assumptions, the average extra income per hectare, where fodder is switched to energy maize, amounts to € 113.8; this is a more than 10% increase in income. This result goes along with a probability that the average extra revenue per hectare is not negative—meaning that the farmer's income will not decrease—of 82.6%. Monte Carlo sensitivity analysis shows that three variables account for the total variance of the farmer's extra income. (i) When the most likely yield of energy maize per hectare is 60 ton fm ha^{-1}, the income of the farmer has a chance of more than 80% to be sustained and even increased by growing energy maize. This result is based on the probability to have a yield that is 30% higher, given equal prices for fodder and energy maize. (ii) When energy maize is sold and fodder maize bought at the minimum price for fodder maize (€ 24 ton^{-1} fm ± 10%, assuming a correlation of 0.5 between the prices), the extra revenue per hectare per year remains positive (€ 65.8 ha^{-1}year^{-1}), still with a Prob(Z > 0) of 75%. (iii) Finally, the assumption that prices of fodder and energy maize are correlated and can be discarded amongst others due to contamination resulting from the uptake of metals. To have an almost certain positive average extra income (Z), differences in prices (€ ton^{-1} fm) for energy maize and fodder maize vary between −0.8 and +2.5.

When extra income can be combined with income from energy production, plant-based technologies can become yet more appealing, given the upsurge of fossil energy prices. Therefore, in the second part of this chapter the potential profitability of a digestion installation fed with metal enriched biomass was investigated. The investment in a digester can be done by a group of cooperating farmers. Cooperation between farmers can be successful, as already shown in Denmark (Raven and Gregersen, 2007). According to our analysis, the minimum number of farmers to have a fair chance that the digesting is profitable is 15, as economies of scale apply for the CHP engine from thereon. Pessimistic scenarios show the importance of the level of the maize price, operational subsidies in the form of GC Certificates and the selling price of electricity produced. Changing each of them with 10% *(ceteris paribus)* renders the installation economically unviable. Each of them is determined by the market and as such not under control of the farmer. However, given current prices of fodder maize, the farmer can decide whether he will sell his energy maize at a given calculated price to a digester and whether or not he is prepared to take part in the digestion project. Concerning the metal enriched digestate, to reach a probability of 90% that the extra income resulting from digestion is positive, it is necessary to find a useful use for the net heat. Not doing this will render the installation unviable. The total average extra revenue for the farmer (*i.e.,* from both growing energy maize and performing digestion) amounts to € 191 ha^{-1} year^{-1}, which means an increase of 17% compared to his income in 2005. Moreover, the probability that this extra income is positive is 90% when he grows energy maize instead of fodder maize and simultaneously exploits the digester in cooperation with other farmers. As such, ecological benefits originating from plant-based technologies go hand in hand with economic benefits for the farmer.

APPENDICES

A. Transport distances

On the regional level it is decided by the Flemish government that 60% of the digestion input has to be related to agriculture, to be allowed to build the digester in an agricultural area (Flemish Government, 2006). A digester fed with energy maize originating from the contaminated area is thus permitted. For now, we assume that transport costs are an income for the seller of energy maize and a cost for the buyer of fodder and energy maize. Each farmer dedicates half of his land to growing maize. The maize however, has to be transported over a distance that covers the whole land (40 ha per farmer). Therefore, to calculate the distance, the number of hectares with maize has to be multiplied by 2. In Figure 4-20 we assume that the digester is geographically located in the centre of the region occupied by the cooperating farms (indicated by the two grey concentric circles in the middle). Energy maize sold by the farmers has to travel an average two-way distance (D_1) to the centre of the circle formed by the cooperating farmers, where the digester will be installed. Fodder maize bought by the cooperating farmers has to travel a longer average distance ($D_2 = (a+b)/2$). It has to come from the uncontaminated area (the grey circle surrounding the contaminated zone), outside the polluted area of 280 km^2 (indicated by the area within the dotted lines).

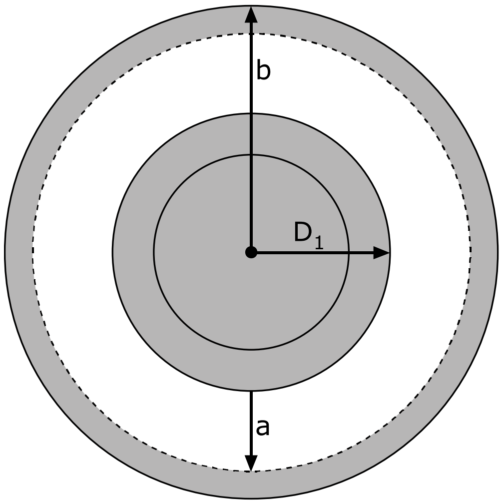


Figure 4-20 Calculation of transport distance for energy maize D_1

A simplified calculation of the transport distance for energy maize D_1 is presented by the following equation:

$$D_1 = ([A \cdot 2/(100 \cdot \Pi)]^{1/2}+0)/2 \cdot 2$$

$$D_1 = (A \cdot 2/(100 \cdot \Pi))^{1/2} \qquad \text{(Eq. 35)}$$

A is the number of hectares grown with energy maize, D_1 is an average two-way distance. Total transport is revenue for the seller of energy maize (cooperating farmer), a cost for the buyer of the energy maize (digester). The distance for fodder maize is given by D_2. Here again, an average distance is calculated, but not a two-way distance:

$D_2 = [(280/\Pi)^{1/2} - (A \cdot 2/100 \cdot \Pi)^{1/2} + [(280/\Pi) + (A \cdot 2/100 \cdot \Pi)]^{1/2}\ ]/2$
$D_2 = [a+b]/2$
$D_2 = [(9.44-D_1)+(89.13+D_1^2)^{1/2}\ ]/2$ (Eq. 36)

B. Calculation of the yearly extra income per hectare

The extra yearly income for the farmer is obtained by recalculating the NPV to an annuity, *i.e.,* a yearly constant CF, which after discounting, would again lead to the NPV. This annuity is calculated by multiplying the NPV with an annuity factor (AF) in (Eq. 37) (i = discount rate = 6%). The annuity is then divided by the number of hectares, resulting in the extra yearly income (over 20 years) per hectare from the digestion activity. Given i = 6% and n = 20, AF = 0.087. We use this formula for the calculation of the capital cost of the CHP engine and digester (Table 4-42) and for calculation of Z.

$$\mathrm{AF} = \frac{i}{1-(1+i)^{-n}} \qquad \text{(Eq. 37)}$$

C. Digester and CHP engine: dimension and investment costs

Based on the assumption that each farmer grows energy maize on 20 ha, with a yield of 60 ton fm per hectare, the dimension of the digester will only depend on the number of farmers that cooperate. The dimension can be calculated according to the following formula (Timmerman *et al.*, 2005; Lemmens *et al.*, 2007).

$Dig = (A \cdot BP_{EM})/365 \cdot Res$ (Eq. 38)

With:
Dig: dimension of the digester (m^3)
A: number of ha (= number of farmers (N) $\cdot$ 20 ha/farmer)
BP_{EM}: yield of energy maize per ha
Res: residence time of the biomass in the digester (days)

If 15 farmers cooperate, the total biomass available is 18,000 ton fm per year. With a residence time of 38 days (dry digestion), a digester of 1,874 m^3 is needed. The number of farmers willing to start the cooperation or to deliver maize as a feedstock to the digester is important.

The life span of the digester is 20 years, with a digressive depreciation scheme and a zero end value (Maeng *et al.*, 1999; Murphy and McKeogh, 2006). The investment costs of the specific installation (Table C1) used in this study come from Goossens (2007), personal communications with experts and own calculations. We will not assume economies of scale, following Timmerman *et al.* (2005) and Lemmens *et al.* (2007). Generalizations should be made with caution as numerical values in literature differ largely due to different assumptions regarding the biomass used, the involved machinery, the included CHP engine, the size of buildings, whether the farmer himself takes care of the construction, *etc.*

Table C1: Specific investment costs digestion installation

Digester parts	*Investment*	*Allocation variable*
Basic installation	97.50	€ ton^{-1} fm biomass
Dewatering installation	10.00	€ ton^{-1} fm biomass
Buildings	28.33	€ ton^{-1} fm biomass
Desulphurization	0.032	€ m^{-3} gas
Measuring, cooling	0.035	€ m^{-3} gas

The produced gas will be burned in a gas engine with heat recovery in a CHP engine. Heat and electricity produced by the engine are calculated respectively as follows:

$$\text{Heat produced} = BP_{EM} \cdot A \cdot G \cdot EV \cdot n_{th} \quad \text{(Eq. 39)}$$

$$\text{Electricity produced} = BP_{EM} \cdot A \cdot G \cdot EV \cdot n_{e} \quad \text{(Eq. 40)}$$

With:

$BP_{EM} \cdot A$: total amount of biomass available (BP_{EM} = 60 ton per ha, A = 20 ha $\cdot$ N)

G: energy value of biomass digested (190 m^3 gas ton^{-1} fm maize)

EV: energy value of gas produced (53% CH_4 leads to 5.3 kWh m^{-3} gas)

n_e and n_{th}: electric and thermal efficiency of the engine (n_e = 41% , n_{th} = 43%)

Each participating farmer dedicates 20 ha to growing energy maize for the digester. The output of heat by the digester is therefore much larger than the demand by the local farmer. The approach in this paper is to consider the amount of biogas produced using the biomass offered by the group of farmers willing to participate in the project. Other demand for heat has to be sought, *e.g.*, district heating, heating of a nearby hospital or swimming pool, and drying and/or processing the digestate. For electricity too, contracts have to be concluded to deliver electricity to a swimming pool, a building with a large electricity demand, a factory, *etc*., or it can be put on the electricity grid. In this paper, all net heat (*i.e.*, after process use) is used to dry the digestate, all net electricity (*i.e.*, after process use) is put on the grid. The dimension of the engine (Dim) in kWe can then be calculated [h_t: the theoretical working hours of the engine assumed per year (7,500 hours)]:

$$\text{Dim} = \text{Eq } 5 / h_t \qquad \text{(Eq. 41)}$$

Investment costs (I_m) of the engine are calculated according to Stroobandt (personal communication, 2007) and Goossens (2007), based on data from specific cases, as follows:

$$I_m = (-386.1 \cdot \ln(900) + 3{,}170.5) \cdot 1.2 \cdot \text{Dim} + \text{Inv elec wiring}; \qquad \text{(Eq. 42)}$$

in case Dim > 900 kWe

$$I_m = (-386.1 \cdot \ln(\text{Dim}) + 3{,}170.5) \cdot 1.2 \cdot \text{Dim} + \text{Inv elec wiring}; \qquad \text{(Eq. 43)}$$

in case Dim < 900 kWe

The life span of the gas engine is 10 years, so a second investment is needed in year 11 to be able to perform the analysis over 20 years. Like the digester, the engine is digressively depreciated.

D. CHP certificates

The support for exploitation coming from CHP Certificates depends on several factors. The CHP system has to be "qualitative", meaning that the Relative Primary Energy Savings (RPE) have to be larger than 0% for units smaller than 1 MW and larger than 10% for larger units. For small units this means that less primary energy should be used than when electricity and heat are produced separately. This is done by comparing the thermal and electrical efficiency of the engine used with European standard values, established in a Ministerial Decision, see (Eq. 44) (CHP Reference Decision, 2006). If this condition is fulfilled, then support (RCHPC) is given, calculated in (Eq. 45). The issue of certificates is assured by the government during 10 years. After the fourth year, the revenues out of certificates will be diminished as a loss of efficiency is assumed. Therefore, RCHPC is multiplied with X. The formula for X is given by (Eq. 46) (CHP Decision).

$$RPE = 1-1/(n_e / REF_e + n_{th}/REF_{th}) \quad \text{(Eq. 44)}$$

$$RCHPC = [1/ REF_e + n_{th} /(REF_{th} \cdot n_e) - 1/ n_e] \cdot Dim \cdot h_w \cdot (1-p_e) \cdot PCHPC/1{,}000 \quad \text{(Eq. 45)}$$

$$X = 100 \cdot (RPE-0.2(t-48))/RPE \quad \text{(Eq. 46)}$$

In (Eq. 44), REF_e and REF_{th} are the European standard electrical and thermal efficiencies as found in the Ministerial Decision. In (Eq. 45), Flemish reference efficiencies are used for calculating RCHPC. These are determined in the CHP Decision (2006). The price of the certificates (PCHPC) is minimum € 27 MWh^{-1} and maximum € 45 MWh^{-1}. In (Eq. 46), t = total months after the past year, in the fifth year for example, t = 60.

Chapter 4.3 *Farm based model*

Abstract

Commonly discussed in research on phytoextraction is the time for a particular crop to remediate a contaminated site. In this chapter we add to this the impact on an economic level of different green remediation strategies. More specifically, one of the conditions in searching for sustainable land use and -management opportunities is that the NPV(AGI) would not decrease (as compared to the situation before remediation). To test the economic viability of plant-based technologies for the management of a moderately trace element enriched field, we developed an economic decision tool based on the Campine case study. The decision tool determines for a farmer with an average of 36 ha farm land what combination of remediation and traditional crops will guarantee him the highest benefit or lowest cost over a 40 year time period, with a general focus on sustainable land use and -management but with actual remediation being the main focal point. The decision model presented in this chapter confirms that phytoremediation works if only physical data are considered. If these physical data are linked to economic data, results become somewhat more complicated. A first important conclusion is that phytoremediation is in any case more cost-effective than conventional remediation, within limited contamination ranges. A second important conclusion is that SRC of willow shows the physical and economic (if combined with HI crops) potential to sustainably remediate moderately Cd enriched land within a period of 40 years, with calculations showing increases in NPV(AGI) for the farmer (as compared to the current situation) at low levels of contamination. A third conclusion that can be drawn from the analysis is that remediation will at any case cost money, but with the correct incentive government can guide the intensity of remediation.

4.3.1 Introduction

A first drawback of the model in Chapter 4.1 is that it does not take into account control costs. As we might expect, these are higher for food threshold values than for soil standards. Hence we see in Flanders a current practice of applying soil standards before food threshold values. This only serves as a first criterion to determine whether food threshold values will be checked. A second drawback of the optimization model lies in its applicability. The model optimizes for each Cd concentration in soil (C_0) the income over infinity. The infinite time line might not appeal to farmers. Moreover, a different approach for each of the hectares does not take into account the economies of scale, *i.e.* cultivation costs will be relatively higher for 1 hectare land than for 18 hectares of land with the same crop. A third drawback relates to the fact that the model is so theoretic that it might just be hard to understand what the actual implications for a farmer might be.

Moreover, the previous chapter already showed that given the contamination range (12-0 mg kg^{-1}), the model only suggests plant-based management of contaminated land in 15% of cases. In general, energy maize is chosen for ranges of Cd concentration in soil between 12 and 10.1 mg kg^{-1}, after which maize is grown (maximum allowed soil concentration is 10 mg kg^{-1}). Willow is chosen between 3.9 and 3.5 mg kg^{-1} until 3.4 mg kg^{-1}, after which asparagus can be grown, and from 0.6 mg kg^{-1} to remediate until 0.5 mg kg^{-1} from which moment endive is grown. Rapeseed is only in competition with energy maize. Energy maize is the preferred crop for large distances to target (DTT), *i.e.* the difference between C_q and C_0, while willow is preferred for average to small DTT.

For all these reasons, we developed a simplified model that, given the limited time period of 40 years[109], calculates maximum DTT, and within these DTT, maximizes NPV(AGI) per DTT, instead of per C_0. This model analyzes whether phytotechnologies can become part of sustainable site cleanup together with conventional remediation technologies, *i.e.* be used in a more acceptable approach with gradual integration of alternative crops in the crop scheme. The model uses one C_q, and one AGI_{HI}. The model starts from a crop scheme, and then determines how many hectares in this crop scheme could be grown with willow for the farmer still to have an income comparable to his income before remediation. Drawbacks of this second model are that the model is not optimizing, it chooses the best of given options. Moreover, the model assumes

[109]In the first model there is no time limit, since in that model the restriction is conceived as such that at every point of Cd concentration in soil, the farmer would only grow crops that are allowed, so that there is no damage.

that remediation is not necessary and maize can always be grown (while this is only the case for Cd concentrations below 10 mg kg^{-1}). This makes the model only applicable for land with C_0 < 10 mg kg^{-1}. Finally, due to the restriction of 40 years, only small DTT can be covered by the model.

The results of both models are hard to compare since we used average values for AGI_{HI} in the second model, while in the first model AGI_{HI} depends on C_q. We used the average in the second model because we are working independent of absolute Cd concentration levels. Moreover, based on the sensitivity analysis results on harvesting willow leaves in the first model, we harvest leaves in the base case of this second model.

4.3.2 Data and methods

4.3.2.1 Physical assumptions

The model approaches the problem from a different perspective than the model in Chapter 4.1. This new model determines for a farmer with an average of 36 ha farm land what combination of alternative and traditional crops will guarantee him the highest benefit or lowest cost over a 40 year time period, with a general focus on sustainable land use and management but with actual remediation being the main focal point. This model, which we will refer to as the "farm based model", is based on the MIP project (see 3.1.1) and Vangronsveld *et al.* (2009), and is built on several assumptions:

(i) An initial crop distribution of alternative (remediation) crops is combined with a traditional crop (fodder maize).
(ii) The reference situation (before remediation) is based on fodder maize.
(iii) The number of hectares of short rotation coppice (SRC) of willow is constant over time.
(iv) Remediation of all hectares from year 1 is not obligatory. Traditional crops are allowed within the initial distribution.
(v) The time range is 40 years.
(vi) When the desired level of contamination has been reached (and on uncontaminated land), an average HI crop is grown.
(vii) The area surface is 36 hectare.

Assuming a predefined time period for remediation, a Cost-Benefit approach determines the projected income from the remediation function, based on the contaminant removal or performance of the remediation crop, soil conditions, and income of the remediation crop and HI crop.

Alternative crops are SRC of willow, rapeseed, and energy maize. Energy maize (*Zea mays*) is a crop with low metal uptake capacities, but has the advantage to produce a high biomass yield (leading to a moderate absolute extraction), and the local agricultural sector is very familiar with this crop, which is cultivated in much the same way as fodder maize. Table 3-1 presents an overview of Cd concentrations and biomass yield for energy maize. Winter rapeseed (*Brassica napus*) can only be grown once every three (four) years and will therefore be grown in rotation with energy maize (1 year of rapeseed followed by 2 years of energy maize). As opposed to maize, the crop is not commonly accepted by Flemish farmers. Table 3-2 presents an overview of Cd concentrations and biomass yield for rapeseed. Only recently, experimental plantings of SRC of willow (and poplar) have occurred on farm land and on the experimental field. Table 3-3 presents an overview of Cd concentrations and biomass yield for SRC. In the base case, both leaves and twigs are harvested.

The traditional crop is fodder maize. In the region Balen, Lommel, Overpelt, and Neerpelt dairy cattle farming is the most important activity, with farmers growing fodder maize and temporary grass as feed for the winter period (61%). The other (main) activity is cereals (31%) (Table 3-7) (FOD Economy: SMEs, independent Professions and Energy, personal communication, October 2009). The reference situation is conceived as a situation where the farmer has basic farming equipment, but will have to buy additional equipment or hire external labor to grow SRC of willow and rapeseed. This will be taken into account through adaptations in the income of these crops.

The model checks every year whether the number of hectares of the highest accumulating crop is the same as at the start in year 1. Although authors claim that in some cases we should use the term phytomanagement instead of phytoextraction (Vangronsveld *et al.*, 2009; Thewys *et al.*, 2010a), this does not take away the fact that in some cases in the (very) long run the goal could/should be functional repair of the soil (Vangronsveld *et al.*, 2009). By making sure that the best remediating crop has a constant part within the crop scheme, the latter goal remains within reach. The choice for SRC of willow can be justified by the fact that SRC of willow has the best remediating capacities among the studied crops (Table 3-3). More specifically, when one hectare of SRC of willow reaches the desired end contamination level, and subsequently a HI crop is grown on this hectare, then a hectare of fodder maize is replaced by SRC of willow from the following year on.

To assess the efficiency of phytotechnologies in reducing risk from pollutants, several factors can be used to express element accumulation in plants.
- bioconcentration factor (BCF) = total element concentration in plant tissue ÷ total element concentration in the contaminated matrix (or the labile pool);
- accumulation factor (AF) = total element amount in plant tissue ÷ total element amount in the contaminated matrix; and
- translocation factor (TF) or shoot-to-root ratio (S/R) = total element concentration in shoot tissue ÷ total element concentration in root tissue (Mench *et al.*, 2010).

The model presented in this chapter is built on linear extraction of metals by crops since (i) the Campine soil is a sandy soil, and chemical equilibriums restore fast when available metal pools are exhausted, and since (ii) we have no long-term data available on the relation between extraction of metals and soil concentrations. However, we do acknowledge that extrapolation of results from short-term experiments based on constant metal accumulation may lead to underestimation of the phytoextraction duration.

Based on the experimental fields installed in the region, Vangronsveld *et al.* (2009) confirm that 5 mg kg^{-1} soil is the mean concentration of Cd found on the experimental field.

Table 4-49 Extraction of Cd to reduce concentration in soil from 5 to 2 mg kg^{-1} (DTT=3) for energy maize, rapeseed, and willow, based on base case values

	Biomass *(ton dm ha^{-1})*†	*Cd* *(mg kg^{-1} dm)*†	*BCF*‡	*Cd removal* *(kg ha^{-1} y^{-1})*	*Cleanup* *time (y)*§
EM	20	1.08	0.22	0.022	880
RS	5.2	2.68	0.54	0.014	999
SRC-twigs	4.8	25	5	0.120	160
SRC-leaves	1.2	40	8	0.048	
SRC-total	6	28	5.6	0.168	114

EM=energy maize, RS= rapeseed, SRC= SRC of willow; †Based on Table 3-1, Table 3-2, and Table 3-3; ‡bioconcentration factor; §calculations based on 40 cm soil depth for energy maize, rapeseed, and willow, a soil density of 1.6 ton m^{-3}, and assuming linear extrapolation

The assumption that remediation is not obligatory has been introduced to make the remediation more easily acceptable for the farmer. The traditional crop (fodder maize) is not assumed to contribute to the remediation of the enriched soil. The time range of 40 years is based on the average economically active life of a farmer, and reduces the initial distance to target (DTT) (Vangronsveld *et al.*, 2009) (Table 4-49).

Start levels of contamination are equal for all 36 hectares. Soil remediation standards, *i.e.* soil concentrations for trace elements from which remediation is obliged, are determined on a regional level (VLAREBO, Flemish regulation on soil remediation and soil protection). However, for historical contamination, no Flemish soil remediation standards (BSN) exist. Nevertheless, in accordance with OVAM (Public Waste Agency Flanders) practices, BSN are also one of the criteria for detecting serious threats in case of historical soil contamination. We focus on Cd, since BSN for Cd (2 mg kg^{-1} dm) are exceeded (VLAREBO). In that same VLAREBO, the target value for Cd in soil is 1.2 mg kg^{-1} dm, *i.e.* the soil concentration we should aim for. Aside from these soil standards, there are also European and Belgian product threshold values. As mentioned in Section 2, they do not seem to coincide with each other and moreover, they are applied inconsistently. Therefore, in this model, we assume that when Flemish soil target values are reached, all vegetables are allowed. The model includes one average vegetable (HI crop) with an average yearly income. The assumption of 36 ha is based on the average size of a Campine farm.

After harvest, crops used for remediation of soil pollution need alternative application. In this chapter we have chosen for energy conversion as a sustainable alternative because energy production will more likely get public approval. In Europe, farmers are variously rewarded for direct positive contributions to biological diversity, improvements to water quality and increased soil health. Many Member States also support bioenergy programs. Moreover, there has already been research on tracing metals in energy conversion installations, while until now, there has been no research yet on tracing metals in other biomass using technologies (such as paper mills). An overview of studied energy conversion combinations can be found in Table 3-4.

4.3.2.2 Economic assumptions

The economic decision tool calculates for one farm (36 ha) the NPV of the Adapted Gross Income (AGI) over 40 years, for different phytoremediation options. For reasons explained in Chapter 3.1 we calculate with the adapted gross income (AGI). AGI's are adjusted for the extra cost as compared with uncontaminated biomass resulting from the enrichment with trace elements, such as disposal of final ashes. The economic impact of metals on the end use of rest products is based on current legislation, and this impact is then calculated back to the price of the original biomass and the resulting AGI for the farmer. Moreover, remainders on the field (such as leaves from SRC of willow) will be collected, and we use best case values (Table 4-17). This results in final r' for the remediating crops as presented in Table 4-50, based on Table 4-2.

Table 4-50 Yearly r' (€ ha^{-1}) of alternative crops for different conversion options

	EM	*RS-PPO-pers use*	*RS -PPO-tractor*	*RS - Biodiesel*	*SRC-electr.*	*SRC-heat*	*SRC-CHP*
r	1,042.68	1,209.44	1,050.06	878.00	110.26	110.26	110.26
r'	1,131	1,272	1,112	940	160	160	160

The traditional crop, fodder maize, has an AGI of € 960 ha^{-1} $year^{-1}$, when sold on the market[110], and assuming that fodder maize grown on the land does not accumulate trace elements, and that as such there is no economic effect on the price of fodder maize. The AGI for an average high income (HI) crop, corrected for rotation with energy maize is € 1,747 ha^{-1} $year^{-1}$.

Based on these physical and economic assumptions, and using an interest rate of 4%, we arrive at a NPV(AGI) over 40 years for a whole farm land of 36 ha. By changing the initial rotation scheme, we are then able to optimize NPV(AGI) (within the defined assumptions), and thus the income of the farmer for different DTT's, different biomass yields, different trace element removal, different AGI's for the different crops, *etc*.

4.3.3 Results

4.3.3.1 Base case

The amount of soil to be treated per farmer is 230,400 ton (Eq. 47). The maximum contamination that the fastest remediation crop (SRC of willow in our case) can remove within 40 years, given base case physical assumptions, is 6.72 kg ha^{-1} (Eq. 48). This translates into a maximum DTT of 1.05 mg kg^{-1} soil between begin and end level of Cd concentration in soil (Eq. 49). Since we assume linear extraction it does not matter whether the model has to remediate from 5 to 3.95 or *e.g.* from 3.05 to 2 mg kg^{-1} soil.

$$S = A \cdot d \cdot \rho \qquad \text{(Eq. 47)}$$

$$Q_i = (Q_q - Q_0) = (BP_i \cdot E_i) \cdot t_i / 1{,}000 \qquad \text{(Eq. 48)}$$

$$DTT = (C_q - C_0) = 1{,}000{,}000 \cdot Q_{max} / S \cdot 1{,}000 \qquad \text{(Eq. 49)}$$

The total amount of soil (S) is calculated as the product of soil depth (d, expressed in m), soil density (ρ, expressed in ton m^{-3}), and area surface (A,

[110]AGI's do not include revenues from the continuation of dairy cattle rearing. Therefore, the model does not include the fact that fodder maize has to be bought from outside the contaminated area to continue this activity. An extended analysis of the latter approach can be found in Chapter 4.2 (Thewys *et al.*, 2010a, b).

expressed in m²). (Q_q – Q_0) is the amount of Cd (kg ha^{-1}) removal and is calculated as the product of the remediation duration (t_i) of crop i in years multiplied with the product of yearly biomass potential of crop i (ton ha^{-1} $year^{-1}$) (BP_i) and concentration of Cd in crop i (mg kg^{-1}) (E_i). Given t_i = 40, we are able to calculate Q_{max}, *i.e.* the maximum amount that crop i can remove within 40 years. DTT is the difference between the concentration of metals initially present in soil of the polluted site (C_0) and the concentration of metals in soil for which it is allowed to grow HI crops (C_q). The maximum DTT (mg kg^{-1}) is calculated as the maximum amount of metals that can be removed per hectare by the fastest remediating crop i (Q_{max}, converted into mg Cd ha^{-1}) divided by the total amount of soil (S, converted into kg soil ha^{-1}).

Table 4-51 Number of hectares that cover the distance to target (DTT) within 40 years given the initial number of hectares of SRC of willow (BASE CASE)

	DTT (mg kg^{-1} soil)						
Willow (ha) in year 1	*1.05*	*0.90*	*0.75*	*0.60*	*0.45*	*0.30*	*0.15*
6	6	6	6	6	12	18	36
8	8	8	8	8	16	24	36
10	10	10	10	10	20	32	36
12	12	12	12	12	24	36	36

The initial crop scheme is a combination of SRC of willow (6 ha), fodder maize (12 ha), energy maize (14 ha), and rapeseed (4 ha) (MIP project, unpublished results). When a hectare of land is remediated (*i.e.* reaches the desired end level of Cd concentration), a HI crop is grown. Moreover, when it is SRC of willow that is replaced by a HI crop, another hectare of willow is added (replaces fodder maize, energy maize and rapeseed in this order), such that the total number of hectares of SRC of willow remains constant over time. Given the base case physical conditions (biomass production, soil density, linear accumulation, and biomass production) (Table 4-49), SRC of willow is able to remove 6.72 kg ha^{-1} within 40 years. Therefore, when the DTT is smaller than 1.05 mg kg^{-1}, SRC of willow reaches the target in less than 40 years, and SRC of willow can be grown on additional hectares. When the DTT is smaller than 0.6 mg kg^{-1}, SRC of willow reaches the target for more than the initial number of hectares of willow within 40 years (Table 4-51).

The above analysis is a general representation of what is commonly discussed in research on phytoextraction: the time for a particular crop to remediate a contaminated site. The added value of this chapter is that it adds the impact on an economic level of different green remediation strategies. More specifically, in our case, one of the conditions in searching for sustainable land management opportunities was that the NPV(AGI) would not decrease as compared to the current situation where fodder maize is grown (MIP project, unpublished results)[111]. This condition is only fulfilled in specific circumstances (Table 4-52). In this table we show NPV(AGI_{rem}) as a percentage of NPV(AGI_{ref}), with NPV(AGI_{rem}) the NPV(AGI) for different distances to target (DTT) and different initial number of hectares of SRC of willow based on AGI during and after remediation, and with NPV(AGI_{ref}) the reference NPV(AGI) based on AGI before remediation. When this percentage is lower than 100%, NPV(AGI) decreases due to the chosen strategy.

Table 4-52 NPV(AGI_{rem})† for different distances to target (DTT) and different initial number of hectares of SRC of willow as a percentage (%) of the reference NPV(AGI_{ref}) before remediation (BASE CASE)

	DTT (mg kg^{-1} soil)						
Willow (ha) in year 1	*1.05*	*0.90*	*0.75*	*0.60*	*0.45*	*0.30*	*0.15*
6	92.81	93.58	94.74	96.21	98.34	103.61	123.55
8	88.18	89.22	90.76	92.72	95.42	102.31	126.74
10	83.55	84.79	86.64	88.98	92.33	102.04	129.03
12	78.92	80.31	82.38	85.00	89.17	102.69	131.52

†40 years, i=4%, 36 ha, base case conditions

When the DTT is equal or smaller than 0.45, there is no negative and even a positive impact on NPV(AGI), due to the HI crops that generate a high AGI and at a faster rate than for high DTT. At higher DTT, it is better to grow less SRC of

[111]Mind that is not the same as demanding that AGI does not decrease in time.

willow, the negative impact on the NPV(AGI) becoming larger at higher DTT's. This can be explained by the fact that at high DTT, SRC of willow is not able to cover the DTT and therefore does not result in higher AGI of HI to compensate for the lower AGI of SRC of willow during remediation. Thus, if the goal is to cover the DTT for the farm land, then, given the worst case scenario of DTT, the best option for the farmer is to grow 6 ha of SRC of willow. This will result in 6 ha remediated land, with a 7% decrease in NPV(AGI) for the farmer.

Instead of the condition of the NPV(AGI) that should not decrease, the question that should be asked might actually be the following: Which remediation strategy should be chosen, a green strategy or a conventional strategy so as to <u>minimize</u> the decrease in NPV(AGI) as compared to the situation before remediation[112]? The decrease in NPV(AGI) can be calculated per remediated hectare A_{rem} (C_{rem} in (Eq. 50)) and per ton remediated soil S_{rem} (c_{rem} in (Eq. 51)).

$$C_{rem} = [NPV(AGI_{rem}) - NPV(AGI_{ref})] / A_{rem} \quad \text{(Eq. 50)}$$

$$c_{rem} = [NPV(AGI_{rem}) - NPV(AGI_{ref})] / S_{rem} \quad \text{(Eq. 51)}$$

If we assume a reference NPV(AGI) of € 684,038[113] over 36 ha, a remediation strategy starting with 6 ha willow[114], and a DTT of 1.05 mg kg^{-1} soil (so that those 6 ha actually cover the DTT), the cost of remediation is in that case € 8,202[115] per remediated ha or € 1.28[116] per remediated ton soil. This is the loss in income due to phytoremediation[117] and is 17 times less than the cost of conventional remediation technologies of € 22.6 ton^{-1} (Table 4-60). This cost of € 1.28 ton^{-1} seems inconsistent with the number found by Vangronsveld *et al.* (2009) (a benefit of € 6.7 m^{-3}), but can be explained by different physical and economic assumptions. We will therefore introduce their data in our model in the sensitivity analysis.

---

[112]Mind that this is a different approach from the previous model where MAC was determined by comparing with clean land.
[113]Reference: fodder maize (36 ha); 40 years, i = 4%
[114]SRC of willow (6 ha), fodder maize (12 ha), energy maize (14 ha), and rapeseed (4 ha)
[115]A_{rem} = 6 ha = 60,000 m²
[116]S_{rem} = 38,400 ton (A = 60,000; d = 0.4; ρ = 1.6)
[117]We assume that fodder maize is still allowed on the land.

4.3.3.2 Sensitivity analysis

<u>Optimistic biomass production and Cd accumulation</u>

Table 4-53 Extraction of Cd to reduce concentration in soil from 5 to 2 mg kg^{-1} for energy maize, rapeseed, and willow, based on Vangronsveld *et al.* (2009)

	Biomass (ton dm ha^{-1})†	*Cd (mg kg^{-1} dm)†*	*BCF*	*Cd removal (kg ha^{-1} y^{-1})*	*Cleanup time (y)‡*
EM	20 (+0%)	3 (+178%)	0.6	0.06	188 (20%)
RS	8 (+54%)	6 (+123%)	1.2	0.05	234 (23%)
SRC-twigs	8 (+67%)	24 (-4%)	8	0.19	117 (73%)
SRC-leaves	2.4 (+100%)	60 (+50%)	2.4	0.14	
SRC-total	10.4 (+73%)			0.34	67 (59%)

Based on Vangronsveld *et al.* (2009); †between brackets the difference with Table 4-49 expressed as a percentage; ‡between brackets the cleanup time as a percentage of the base case cleanup time in Table 4-49

Table 4-53 is taken from Vangronsveld *et al.* (2009). The biomass yield for rapeseed is 50% higher than the average found in literature, which we used in the base case. The average biomass yield of SRC of willow used in the base case is low. This is not surprising as soil conditions in the area are not very favorable (dry, poor and sandy) for growing these species (Ruttens *et al.*, 2008). It is expected that biomass will increase during the next rotation cycles, and 10 ton dm is an average often found in literature. All differences in biomass yield between Table 4-49 and Table 4-53 could be explained by the fact that the crops from Table 4-53 were grown immediately after the last agricultural activity on the field. The alternative crops could therefore benefit from the nutrients still present in the soil due to fertilization by farmers. In the next experiments, no additional fertilizer was added, resulting in lower biomass yields (Table 4-49). The differences in Cd accumulation between Table 4-49 and Table 4-53 could be explained by the fact that data in the former table are based on an average of different species/clones, whereas data in the latter table are based on one specific species/clone. Moreover, accumulation of trace elements is also vulnerable to seasonal conditions, which differ over several years. This results in cleanup times that are 20-73% of the base case cleanup times. The impact of these differences is shown in this part.

Table 4-54 Number of hectares that cover the distance to target (DTT) within 40 years given initial number of hectares of SRC of willow (based on Table 4-53)

	DTT (mg kg^{-1} soil)						
Willow (ha) in year 1	*1.05*	*0.90*	*0.75*	*0.60*	*0.45*	*0.30*	*0.15*
6	12	12	24	36	36	36	36
8	16	22	36	36	36	36	36
10	22	32	36	36	36	36	36
12	30	36	36	36	36	36	36

Based on the new physical data, the maximum contamination that SRC of willow can cover within 40 years, is 13.28 kg ha^{-1} (Eq. 48). This translates into a DTT of 1.77 mg kg^{-1} soil (Eq. 49). If initially 6 ha of SRC of willow are planted, then 36 hectares are remediated within 40 years, given that the DTT does not exceed 0.60 mg kg^{-1} (Table 4-54). Or, if the DTT is 0.90, 36 ha can only be remediated within 40 years if 12 ha of SRC of willow are initially planted.

Table 4-55 NPV(AGI_{rem})† for different distances to target (DTT) and different initial number of hectares of SRC of willow as a percentage (%) of the reference NPV(AGI_{ref}) before remediation (based on Table 4-53)

	DTT (mg kg^{-1} soil)‡						
Willow (ha) y 1	*1.05*	*0.90*	*0.75*	*0.60*	*0.45*	*0.30*	*0.15*
6	100.81	102.37	105.80	113.19	121.95	136.61	152.82
8	98.17	100.44	106.50	115.26	124.17	138.20	155.33
10	95.56	99.05	106.96	115.75	125.30	139.86	156.91
12	93.17	98.02	106.84	115.64	125.38	140.81	157.95

†40 years, i=4%, 36 ha; ‡If for the impact on NPV(AGI) we would use 1.77 mg kg^{-1} as DTT instead of 1.05, and 1.52 instead of 0.9 (=0.9/1.05 · 1.77), *etc.*, this would result in a table identical to Table 4-52. To be able to compare with Table 4-52, we therefore use equal DTT's.

When DTT is 0.75 mg kg^{-1} or lower, there is always a positive impact on the NPV(AGI) due to the HI crops that generate a high AGI, and at a faster rate than for high DTT. It is thus better to initially grow as much SRC of willow as possible. In general, at higher DTT, it is better to grow less SRC of willow. Thus, if the goal is to cover a DTT of 1.05 mg kg^{-1} for the farm land, then the best option for the farmer remains growing 6 ha of SRC of willow (as in the base case). This will now result in 12 ha remediated land (Table 4-54), with a 0.8% increase in NPV(AGI) (Table 4-55). Again, if we assume a NPV(AGI_{ref}) of € 684,038 over 36 ha, and a remediation strategy starting with 6 ha willow, and a DTT of 1.05 mg kg^{-1} soil, the cost of remediation is now € -927 per remediated ha (Eq. 50) or a revenue of € 0.12[118] per remediated ton (Eq. 51).

Willow harvest without leaves

If SRC of willow is harvested without leaves, this has an impact on cleanup time and on AGI. The maximum DTT is 0.75 mg kg^{-1}, and the additional income from harvesting and combusting leaves is foregone, a loss of € 50 ha^{-1}, resulting in an AGI of € 111 ha^{-1}.

Table 4-56 Number of hectares that cover the distance to target (DTT) within 40 years given initial number of hectares of SRC of willow when leaves are left on the field

	DTT (mg kg^{-1} soil)				
Willow (ha) in year 1	*0.75*	*0.60*	*0.45*	*0.30*	*0.15*
6	6	6	6	12	36
8	8	8	8	16	36
10	10	10	10	20	36
12	12	12	12	24	36

In any case, 36 hectares are only remediated within 40 years when the DTT does not exceed 0.15 mg kg^{-1} (Table 4-56). This is because the leaves are responsible for almost 30% of total metal extraction by SRC of willow ($BP_i \cdot E_i$). Leaving the leaves on the field increases remediation duration with 30%, resulting in SRC of willow being grown on additional hectares much later, and this has its impact when the maximum time period is set fixed at 40 years.

[118] S_{rem} = 45,000 ton (A=60,000; d = 0.5; ρ = 1.5)

Table 4-57 NPV(AGI_{rem})† for different distances to target (DTT) and different initial number of hectares of SRC of willow as a percentage (%) of the reference NPV(AGI_{ref}) before remediation (willow leaves are left on the field)

	DTT (mg kg^{-1} soil)				
Willow (ha) in year 1	*0.75*	*0.60*	*0.45*	*0.30*	*0.15*
6	87.65	88.97	90.79	94.59	110.11
8	82.73	84.50	86.91	91.85	112.64
10	77.81	79.97	82.93	89.08	114.85
12	72.90	75.41	78.84	86.46	116.71

†40 years, i=4%, 36 ha

In case SRC of willow cannot be followed by HI crops, the least SRC of willow as possible should be grown. Again, if we assume a reference NPV(AGI) of € 684,038 over 36 ha, and a remediation strategy starting with 6 ha willow, and a DTT of 0.75 mg kg^{-1} soil, the cost of remediation is now € 14,082 per remediated ha (Eq. 50) or € 2.2 per remediated ton (Eq. 51).

<u>NPV(AGI): change in willow and maize price</u>

Changing the income levels will not have an impact on the remediated hectares, given an initial crop scheme.

In the base case, a wood price of € 50 ton^{-1} dm leads to a AGI_W (incl. metal waste disposal and harvesting the remainders on the field) of € 160 ha^{-1} year^{-1}. Raising the price of wood to € 75 ton^{-1} dm leads to a AGI_W of € 272 ha^{-1} year^{-1}. Raising the price to € 100 ton^{-1} dm results in a AGI_W of € 384 ha^{-1} year^{-1}. Compared to the base case, there is a small improvement when the price of wood raises to € 75: a farmer could sustain his income from a DTT of 0.45 mg kg^{-1} (0.30 mg kg^{-1} in the base case). The maximum DTT, such that the NPV(AGI) of the farmer does not reduce, becomes 0.6 mg kg^{-1} in case the price of willow rises to € 100 ton^{-1} dm.

Table 4-58 NPV(AGI_{rem}) for different distances to target (DTT) and different initial number of hectares of SRC of willow as a percentage (%) of the reference NPV(AGI_{ref}) before remediation (willow price of € 75 ton^{-1} dm)

	DTT (mg kg^{-1} soil)						
Willow (ha) y 1	*1.05*	*0.90*	*0.75*	*0.60*	*0.45*	*0.30*	*0.15*
6	94.75	95.53	96.69	98.16	100.28	105.55	125.21
8	90.77	91.81	93.35	95.31	98.02	104.90	128.69
10	86.79	88.03	89.88	92.22	95.57	105.16	131.21
12	82.81	84.20	86.27	88.88	93.05	106.22	133.86

In the base case, a maize price of € 30 ton^{-1} fm leads to AGI_{EM} (including metal waste disposal and harvesting the remainders on the field) of € 1,131 ha^{-1} $year^{-1}$. Lowering the price of energy maize to € 25 ton^{-1}, results in a AGI_{EM} of € 831 ha^{-1} $year^{-1}$. Mind that now AGI_{ref} will also reduce from € 960 ha^{-1} $year^{-1}$ to € 710 ha^{-1} $year^{-1}$. This actually results in an improvement as compared to the base case (Table 4-59). This is because the reference income is 100% based on fodder maize, while the remediating income is only partly based on energy maize and fodder maize. The maximum DTT such that the NPV(AGI) of the farmer does not reduce becomes 0.6 mg kg^{-1} in case the maize price goes down to € 25 ton^{-1} dm.

Table 4-59 NPV(AGI_{rem}) for different distances to target (DTT) and different initial number of hectares of SRC of willow as a percentage (%) of the reference NPV(AGI_{ref}) before remediation (maize price of € 25 ton^{-1} fm)

	DTT (mg kg^{-1} soil)						
Willow (ha) year 1	*1.05*	*0.90*	*0.75*	*0.60*	*0.45*	*0.30*	*0.15*
6	94.18	95.57	97.64	100.25	104.05	113.54	148.75
8	89.88	91.73	94.49	97.98	102.90	115.86	156.98
10	85.58	87.84	91.20	95.46	101.74	119.24	162.97
12	81.28	83.89	87.79	92.71	100.62	123.37	168.70

If we assume a reference NPV(AGI) of € 505,903 over 36 ha, and a remediation strategy starting with 6 ha willow, and a DTT of 1.05 mg kg^{-1} soil, the cost of remediation is € 5,986 per remediated ha (Eq. 50) or € 0.94 per remediated ton

(Eq. 51) in case the price of willow raises with 50%, and € 4,904 per remediated ha (Eq. 50) or € 0.77 per remediated ton (Eq. 51) in case the price of maize reduces with 15%.

4.3.4 Discussion and conclusion

The decision model presented in this chapter confirms that phytoremediation works if only physical data are considered. If these physical data are linked to economic data, results become somewhat more complicated. A first important conclusion is that phytoremediation is in any case more cost-effective than conventional remediation, within limited contamination ranges. Witters *et al.* (2009) already gave a comprehensive overview of different conditions under which phytoremediation should be preferred over conventional remediation, but extended economic calculations had not yet been made to date.

Given one hectare of land with a DTT of 1.05 mg kg^{-1} (since this is the maximum DTT the phytoremediation technology can cover in the base case and makes comparison with conventional remediation possible): should one best grow SRC of willow (most effective[119] option) or apply conventional remediation? Given base case conditions, conventional remediation cannot compete with SRC of willow (Table 4-60), unless the cost of conventional remediation goes down to € 2.5 ton^{-1} soil (given NPV(AGI_{ref})= €19,001 ha^{-1}), or the AGI of HI crops raises substantially (€ 8,244 ha^{-1} $year^{-1}$, applying (Eq. 52) on € 163,171 ha^{-1}).

[119]Effective= DTT covered

Table 4-60 Cost (€ ha^{-1}, unless otherwise indicated) of effective remediation (DTT covered) using SRC of willow and conventional remediation

	SRC	*Conventional*
NPV(AGI)	+ 3,171†	+ 34,577†
Remediation cost (year 0)	0	-160,000‡
Net benefit (+) cost (-)	+3,171	-125,423
Marginal difference with reference situation‡	-15,830	-144,424
Net Benefit (+) or cost (-) of remediation compared to reference situation (€ ton^{-1}) (Eq. 51)	-2.5	-22.6

†NPV(AGI) of SRC of willow (SRC) or HI crops (conventional) over 40 years; ‡€ 25 ton^{-1} for biological remediation (OVB (Association for entrepreneurs in soil remediation, Belgium), personal communication, February 2010); ‡in the reference situation fodder maize is grown, with NPV(AGI) = € 19,001

The purpose of the model is to reduce this cost of SRC based phytoremediation, by including it in a crop model. By including other alternative crops such as energy maize with a much higher AGI, the average cost of remediation per remediated ha goes down drastically, depending on the crop scheme (in the base case, with a DTT of 1.05 mg kg^{-1} and starting with 6 ha willow, the average cost of remediation was € 8,202 ha^{-1} over 40 years, or € 1.28 ton^{-1} soil).

Phytoremediation with SRC of willow has already been applied on a commercial scale in the United States (Licht and Isebrands, 2005), Sweden (Mirck *et al*., 2005), and Denmark (Jensen *et al*., 2009), but its advantage over other crops, and its optimal rotation had not been calculated before. Therefore, a second important conclusion from this study is that SRC of willow shows the physical and economic (if combined with HI crops) potential to sustainably remediate moderately Cd enriched land within a period of 40 years, with calculations even showing increases in NPV(AGI) for the farmer at low levels of contamination, if and only if the optimal cropping scheme as suggested by the model is employed, and within contamination ranges defined by the model.

Notice that the reference situation is very important in calculating the cost of (phyto)remediation. In Table 4-60, we compare the income during and after remediation with the income that is currently earned on contaminated land (<10 mg Cd kg^{-1} soil, fodder maize). What we calculate is the cost to change the current situation. If we would compare with what would have been possible on uncontaminated land, the reference income would be higher and the cost of phytoremediation with SRC of willow would be higher, while the cost of conventional remediation would be € 160,000. What we calculate is the loss in income due to the contaminated situation. This would be the basis for compensation for farmers.

Therefore, a third conclusion that can be drawn from the analysis is that, based on this model, remediation will at any case cost money. Until now no initiative had been taken to actually calculate the cost per hectare and it was and still is generally assumed that these costs should and will be borne by the farmer, with the resulting lack of action by stakeholders and the tolerance thereof by local and federal authorities. Therefore, we developed a suggestion for Flemish government to bear remediation costs: depending on the height of her support for the involved farmers, she decides how fast soils will be remediated, *i.e.* based on a goal based support mechanism.

Table 4-61 Compensation based on average yearly change per ha (over 36 hectares and 40 years) in AGI compared to the reference AGI on uncontaminated land (€ ha^{-1} y^{-1}) when phytoremediation is used to cover different levels of DTT, for different hectares of willow in year 1 (BASE CASE)

Willow	*1.05*	*0.9*	*0.75*	*0.6*	*0.45*	*0.30*	*0.15*
6	-856	-849	-837	-823	-803	-752	-561
8	-900	-890	-876	-857	-831	-765	-530
10	-945	-933	-915	-893	-861	-767	-508
12	-989	-976	-956	-931	-891	-761	-484
14	-1,034	-1,019	-997	-969	-912	-754	-469
16	-1,078	-1,062	-1,038	-1,008	-928	-749	-455
18	-1,123	-1,104	-1,076	-1,040	-940	-739	-444
20	-1,177	-1,150	-1,110	-1,059	-950	-742	-442
22	-1,230	-1,195	-1,143	-1,077	-960	-745	-441
24	-1,284	-1,241	-1,177	-1,096	-970	-748	-440
26	-1,338	-1,287	-1,211	-1,114	-980	-751	-438
28	-1,392	-1,333	-1,245	-1,133	-991	-754	-437
30	-1,446	-1,379	-1,278	-1,151	-1,001	-757	-436
32	-1,500	-1,425	-1,312	-1,170	-1,011	-760	-434
34	-1,543	-1,460	-1,337	-1,180	-1,013	-756	-427
36	-1,587	-1,496	-1,362	-1,191	-1,015	-752	-420

For sites with a DTT between 1.05 mg kg^{-1} and 0.15 mg kg^{-1}, we calculated the average loss in AGI (€ ha^{-1} $year^{-1}$) as compared to uncontaminated soil (NPV(AGI) = € 34,577 ha^{-1}) (average, since our crop scheme includes multiple crops, and losses and surpluses are averaged over the whole farm of 36 ha) Table 4-61 is based on (Eq. 52) where the last term is the annuity factor based on an interest rate (i) of 4% and a time period (n) of 40 years.

$$([NPV(AGI_{rem}) - NPV(AGI_{ref})]/36) \cdot (i/[1-(1+i)^{-n}]) \qquad \text{(Eq. 52)}$$

If conventional remediation would be used for the whole farm, the average yearly support per hectare would have to be € 8,083 for 40 years **(Eq. 52)**. The model demonstrates, by performing different sensitivity analyses representing various conditions, that phytoremediation could be used on moderately contaminated land and has a competitive advantage over conventional remediation. Moreover, it shows that with the correct incentive, government can guide the intensity of remediation.

However, what the model does not do is convince the tenant to remediate, since we assume that, given the currnt soil conditions, the tenant (in this case the farmer) is still allowed to grow fodder maize on the land. The reasons for this approach are three-fold. First, legislation on soil remediation is inconsistently applied in Flanders, the region for which the model was initially developed, resulting in farmers practicing in the contaminated zone being allowed to continue their dairy cattle activities, but no longer being allowed to grow HI crops (vegetables). Second, gradual introduction of phytoremediation crops might contribute to its approval. Therefore, fodder maize will only gradually be replaced and the income on these hectares will be similar to the income on uncontaminated hectares. Third, fodder maize does not accumulate Cd in a way which endangers its use as fodder.

The responsibility to convince land owners to remediate land lies with policy makers, who are greatly depending on studies which demonstrate total benefits (*i.e.* private and social) of remediation. Given current BSN, current food threshold values below which HI crops can be grown safely, and small DTT's that can currently be covered by remediation crops, the model ends up remediating soils with a concentration that lie below BSN. This issue has been addressed in the alternative model in Chapter 4.1 where we argue for the consistent use of legislation. More advanced management procedures for contaminated land are needed, based on several phases: the development of the conceptual model combined with a risk assessment, resulting in a feasible and practical remediation strategy option. Moreover, regulators will also have an important role to play in determining requirements for biomass disposal. After energy conversion or alternative use of the biomass, there still is a significant volume of contaminated biomass for other processes or disposal. Investigations must therefore focus on the long-term possible synergy between energy crop production and contaminant phytoextraction. This issue is covered in Section 3.

Section 5

GENERAL DISCUSSION AND CONCLUSIONS

Chapter 5.1 *Introduction*

"A multidisciplinary approach is warranted to make phytoextraction a feasible commercial technology to remediate metal-polluted soils" (do Nascimento, Xing, 2006).

As a result of past activities of different pyrometallurgical zinc smelters, an area of about 700 km² in the North East of Belgium and in the South of the Netherlands (Campine region) is contaminated with metals (Hogervorst *et al*., 2007). These metals are mainly concentrated in the upper layer of the soil (30-40 cm), which in the Belgian part alone covers 280 km² (www.ovam.be). A large portion of this area is currently in agricultural use. A shift from pyrometallurgical to hydrometallurgical process technologies in the early 1970's drastically reduced emissions of metals to the environment, but historic soil contamination is still responsible for a continued metal exposure of people and environment in the area. Soil Cd concentrations in the region range between values below the background value of 0.7 mg kg^{-1} and above 30 mg kg^{-1} Cd (and several hundreds mg kg^{-1} for Zn and Pb), while background metal levels in these soils lie around 0.7 mg Cd kg^{-1}, 77 mg Zn kg^{-1}, and 31 mg Pb kg^{-1}. In this region, metal mobility in soils is relatively high, due to the sandy texture and an acid soil pH (pH-KCl values range between 5 and 5.5). As a result, several local vegetable harvests (*e.g.* carrot, scorzonera) cultivated for food industry have already been confiscated by the Belgian Federal Agency for Food Safety (FAVV) because Cd concentrations in the crops were exceeding legal threshold values for human consumption, even when total concentrations in soil were only slightly exceeding background values. Moreover, control on food- and fodder safety will only become stricter in the future, leading to even more declarations of non-conformity in the Campine region. Regional policy therefore prescribes that the soils should be remediated, while at the same time it is desirable to keep the income of the farmers stable.

However, the area has such a large extent that conventional remediation is not applicable. Cultivation of crops with moderate to high metal accumulation capacities on metal contaminated soils should allow to gradually reduce soil metal concentrations, in particular the bioavailable fractions. This principle is known as metal phytoextraction. The final goal of the process is to obtain a remediated soil that can again be safely used for food- and fodder production. If produced biomass can be valorised into an alternative income for the farmer, then the main drawback of phytoextraction, the extended remediation period, may become invalid, and slower working phytoremediation schemes based on gradual attenuation of the contaminants rather than short-term forced extraction may be envisaged (Robinson *et al.*, 2003). The utilization of the obtained biomass of a phytoextraction cycle as an energy resource is therefore attractive (Chaney *et al.*, 1997) and could pose an attractive economic alternative to local farmers.

In this context, from 2004, a large-scale (10 ha) experimental field was installed in Lommel (Belgium) to evaluate the possibilities of cultivation of energy crops with low and moderate affinity for metal accumulation as an alternative for farmers growing food and fodder crops on these historically contaminated soils. Energy crops of main interest include short rotation coppice (SRC) of willow (*Salix spp.*) and poplar (*Populus spp.*), energy maize (*Zea mays*) and rapeseed (*Brassica napus*).

We argue for a holistic perspective on the Campine case, because a too narrow focus on one problem at a time can, at best, prevent taking advantage of potential synergetic effects, or, at worst, make another problem even more serious (Berndes *et al*., 2008). Because there are additional externalities resulting from the cleanup, our analysis does not stop at a private cost-benefit analysis, but takes into account these externalities and values their consequences on social welfare. According to Pollard *et al.* (2004, 2008) successful contaminated land management and policy thereon should be based on interdisciplinary knowledge, combining natural, social and engineering sciences. Decision making should be considered as a process, based on participation, communication and deliberation. OVAM (2005) generated a report with an overview of all actors involved in the contaminated soil problem, ranging from governments over civil society, industry and the agricultural sector to nature protection associations, each with an indication of their relation to the problem (victim or solver) and their potential input to the solution. Current regulation should be open for discussion, and economic incentives are primordial. In the future, all different actors would like to come to a general action framework that generates more clarity concerning allowed activities (and more specifically concerning allowed crops), current legislation on threshold values, responsibility, potential solutions, and finances. The two current policies, a soil policy, and a food- and fodder policy (federal and regional), are unclear and inconsistently applied. The system of control on food produced on the land is a trapped system where a first selection is based on soil standards. The basis for both policies is not clear, and control on application of the legislation is inconsistent.

Chapter 5.2 *Research questions answered*

Our resulting question in this study is therefore:

"Does phytoremediation offer a multifunctional and sustainable alternative for conventional remediation technologies for functional repair or management of metal contaminated agricultural sandy soil, resulting in economically optimal remediation strategies using a legislation based business model?"

Can plant-based technologies offer an economically viable alternative for conventional remediation technologies and which crops should be used and under what circumstances? The decisions in this study are based on a cost-benefit analysis. Does phytoremediation result in other than private costs and benefits and if so, are these (externalities) positive or negative? The resulting biomass could be used for (i) renewable energy purposes with a potential CO_2 abatement, but also results in (ii) metal waste disposal. Both are considered externalities of phytoremediation and should be internalized in the decision model through the correct policy. Do current policies internalize in an economically efficient way the externalities of soil pollution? Do current policies correctly internalize the externalities of phytoremediation?

5.2.1 Does phytoremediation result in other than private costs and benefits and if so, are these externalities positive or negative?

Market prices do not always correctly reflect a consumer's Willingness to Pay (WTP). An activity may generate impacts/outputs that spill over to other economic agents when among the multiple outputs generated, there are some that are welfare-enhancing (positive externality) or welfare-reducing (negative externality), but for which no (correct) private markets exist (yet). When agricultural crops are used for both remediation and energy production, these externalities are (+) CO_2 abatement of biomass based energy because the production and use of this energy will avoid the use of fossil fuels, and (-) the elevated presence of metals in the rest product after energy conversion because the presence of metals is not yet reflected correctly in biomass prices. Therefore, in determining the economic performance of the phytoremediation pathway, adjustments need to be made. Different methodologies are available for estimation of the economic value of externalities. These externalities are then internalized through government intervention (policy).

5.2.1.1 CO_2 abatement potential of biomass for energy

In Europe, the production of energy maize is increasing rapidly. The resulting biomass from this crop can be applied for conversion into biogas through anaerobic digestion (Thewys *et al.*, 2010b). Also, calculations on the energy potential of willow in Belgium seem promising (Cidad *et al*., 2003). Rapeseed results in rich oil containing seeds, and straw (FOD Economy: SMEs, independent Professions and Energy, personal communication, March 2009). Moreover, on a European level, the impact assessment of the thematic strategy on soil protection summarizes different positive and negative environmental, economic, and social impacts of soil remediation. The extensive use of energy for the excavation, transport and treatment of contaminated soil is a negative environmental impact of remediation.

From the life cycle analysis (LCA) performed in this study for each of the three studied crops and their conversion technologies we concluded that each of these crops offers the potential to reduce CO_2 emissions. We took into account the marginal impact of the metals in the biomass on the energy conversion efficiency and on the potential use of the biomass and its rest products after conversion. Our analysis shows that digestion of energy maize with combustion of the metal enriched digestate shows the best energetic and CO_2 abating perspectives. Both rapeseed conversion options (Pure Plant Oil (PPO) and biodiesel) have the same "net CO_2-net energy" ratio as energy maize, indicating that both rapeseed options can only become better than energy maize by improving energy efficiency, and biomass yield. Willow combined with Combined Heat and Power (CHP) production has a net energy production that does not result in as much CO_2 abatement as energy maize. It might therefore be concluded that willow might better be used for other options for which the net energy results in a higher CO_2 abatement. An example is the conversion of willow to replace cokes based electricity generation. Although in our case study, this crop-conversion option shows a rather low net energy production potential, its "net CO_2-net energy" ratio shows promising perspectives concerning its potential contribution to CO_2 abatement. This does however not mean that in the long run willow should not be used for combined conversion into heat and electricity, but only after all electricity installations with cokes have been converted into willow (or other biomass) based installations or have been replaced by a CHP turbine.

Given a ratio in biomass yield (dry matter, dm) of 0.3 of willow to energy maize, net energy ratios lie between 0.14 (co-combustion of willow in a cokes based electricity installation) and 0.44 (co-combustion of willow in a new CHP turbine), and CO_2 abatement ratios between 0.33 (combustion of willow in an existing CHP installation) and 0.67 (co-combuston of willow to replace cokes based heat). SRC of willow could reach the same energy yield per hectare per year as energy maize, given that the relative biomass yield is 0.64, translating into a biomass yield for willow of 13 ton dm ha^{-1} $year^{-1}$. This lies above the average yield found in literature, but has been reached by one of the non-commercial INBO (Research Institute for Nature and Forest) clones on the experimental field in Lommel. Moreover, SRC of willow could abate the same amount of CO_2 per hectare per year as energy maize, already when the relative biomass yield is 0.44, translating into a biomass yield for willow of 8.7 ton dm ha^{-1} $year^{-1}$.

Both net energy potential and CO_2 abatement can be economically valued, with the former being a private benefit, while the second an external benefit, and both have an impact on optimal crop choice. Both benefits do not necessarily go hand in hand: crop-conversion options with an at first sight low net energy production could contribute substantially in our fight against climate change when we consider their potential CO_2 abatement.

5.2.1.2 Metal contaminated biomass

Most literature on the technology of phytoremediation is fundamental, *i.e.* studies the process of metal uptake and translocation itself, and merely mentions the external effects (if even named as such), as a focus for further research. Dushenkov *et al.* (1995) state that the commercialization of rhizofiltration will be driven by economics, by technical advantages, by the reduced volume of secondary waste, the possibility of recycling, and the likelihood of regulatory and public acceptance. The EPA (2000) points to the fact that metal accumulating plants will need to be harvested and recycled or disposed off according to applicable regulation. This is also confirmed by Sas-Nowosielska *et al.* (2004) and Ghosh and Singh (2005). Garbisu and Alkorta (2001) state that harvestable parts can be easily and safely processed (drying, composting, ashing). Dickinson and Pulford (2005) mention risks to the ecological food chains and to the wider environment during crop growth and after the energy conversion process. However, they perceive these risks as either largely insignificant or manageable. There is little literature concerning research on (the actual economic valorization of) externalities resulting from biomass production on contaminated land. Salt *et al.* (1998) mention metal recovery as an option. Borjesson (1999b) and Berndes *et al.* (2004) point to the fact that Cd in harvested willow stems needs to be collected and deposited in a safe manner.

More than 20 years ago, Nelson *et al.* (1987) stated that there will always be a need to dispose off residues in the environment because some toxic constituents cannot be destroyed, or are not commercially interesting. Therefore, the volume of hazardous waste should be minimized and disposal technologies should be optimized to prevent negative environmental consequences. Metals are an example of stock pollutants, they accumulate over time and environment has little or no absorptive capacity for them, as opposed to flow or fund pollutants. Stock pollutants can create a burden for future generations by passing on a damage cost which persists well after benefits received from incurring that damage cost have been forgotten (Tietenberg, 2003). It is not possible to destroy metals, but we should aim to find a solution so that they do not pose a threat to current and future generations. Turner (2000) criticizes the obliged order of the waste management hierarchy as defined in the European Union (and the Ladder of Lansink in Flanders for that matter). He suggests that we use a cost-benefit analysis (preferably accompanied by a LCA) and choose the option that generates the largest social benefit instead of following the hierarchy blindfolded. This is because properly designed and implemented economic incentive instruments allow any desired level of pollution cleanup to be realized at the lowest overall cost to society. They provide incentives for agents who are able to reduce pollution at reduced costs to contribute more to the final goal than other agents (Hahn and Stavins, 1992). Rather than equalizing pollution levels, economic incentive based approaches equalize marginal abatement costs. Overall efficiency is then achieved when the marginal cost of control is equal to the marginal damage caused by the pollution. We offer a systematic approach to find a solution for biomass coming from plant-based technologies for contaminated land management.

Since elevated metal concentrations in soils cause increased metal concentrations in plants, and might therefore lead to substantial health risks to man and environment, we consider this elevated presence of metals in the harvested biomass as an externality of phytoremediation. A crucial aspect when growing crops on contaminated soil is the fact whether the harvested crops will be classified as (hazardous) waste, or as biomass since this has an impact on the further utilization and valorization of the crop. Therefore, we unraveled the multitude of definitions and regulations that are related to the concept of contaminated energy crops, as this might have technical and thus economic implications for conversion installations. We found no evidence of legislation that could exclude energy crops grown on contaminated land from being classified as biomass. We suggest that classification of energy crops with metal accumulating purposes should depend on the main purpose of growing them. If the main purpose is sustainable use of contaminated agricultural land in function of energy production, crops should be considered as biomass. Phytoremediation can be denominated as the main purpose when it involves crops which are capable of (hyper)accumulating metals such that concentrations in soil decrease substantially (such as with hyperaccumulators). Given the relatively low accumulation potential of the crops under investigation, we conclude that the main purpose is sustainable land use and management, with energy production for alternative income generation being the main purpose. For wood waste, the distinction might actually become important for conversion, as different legislation will apply depending on classification, having implications on types of combustion installations and emission control. For digestion, there is no specific written legislation on maximum concentrations of certain elements in input. When (biomass) waste is anaerobically digested, threshold values from the Flemish Regulation concerning Waste Prevention and –Management (VLAREA) on the output have to be respected. For production of biodiesel and PPO there are no quantified specifications regarding the input. It does not matter whether rapeseed is regarded as biomass waste or biomass, the classification does not restrict energy conversion options. Although for energy maize and rapeseed there is no explicit legislation on input for energy conversion installations, there is a close link with the rest product: the concentration of metals in the input will determine the concentration of metals in the rest product and the latter has to comply with threshold values. This means that we should not consider all biomass resulting grown on metal enriched land and used for energy purposes as equal, but rather analyze the rest product after energy conversion for its contaminant levels.

For the three studied crops we decide for our case study that metals don’t have any marginal effect on the energy conversion (technical) efficiency. In our calculations, the separation and drying technology are available at the digestion installation. We compared combustion of willow with combustion of coal, cokes and other waste and decided that the marginal effect on metal emissions of replacing coal, cokes and other waste with contaminated willow would be negligible. The considered combustion installations have a $deNO_x$ electrostatic filter, a $deSO_x$ installation and a desulphurization installation.

Moreover, no metals end up in the energy carrier, but are concentrated in the rest products: ashes, digestate, and cake. Unsafe use and disposal might lead to substantial risks for human health and the environment in general. The presence of metals in the residue after conversion of biomass is therefore a scholarly example of an externality as this fact is not reflected in the price of the original biomass. We study the impact of the contamination on processing the rest products for secondary use or disposal, since a different end use might have to be found for the rest products, to comply with legislation. By strictly applying the rules, the presence of metals in the biomass is internalized. This is based on the assumption that these rules and threshold values are based on *e.g.* health impact studies.

Strictly applying current legislation, applying part of the digestate on the land and combusting the rest results in the highest total economic value. When digestate is uncontaminated the economically most viable option is maximum use as fertilizer and combustion of the rest. This results in a positive economic value of € 114 ha^{-1} $year^{-1}$. When the same digestate comes from metal enriched energy maize, less digestate can be applied on the land as fertilizer and the rest is combusted. This results in a negative economic value of € 103 ha^{-1} $year^{-1}$. Therefore, the marginal effect of the metals on the value of the rest products based on valuation only through market prices is € 217 ha^{-1} $year^{-1}$. Remarkable but theoretically possible, the presence of metals should not have an effect on the economic value of the rapeseed cake. This conclusion is based on current legislation on maximum allowed concentrations in fodder. For cake, the use as fodder thus results in no marginal impact from metals. Willow ashes have to be disposed off because metal concentrations are too high. For willow too, the economic effect of metals on the economic value of the ashes is negligible. This is because the amount of ashes per hectare is low and the economic value of uncontaminated ashes to be used as granulates is also negligible. Therefore, the presence of metals resulting in disposal of the ashes has almost no marginal effect. The difference in economic value between contaminated and uncontaminated rest product is taken into account in the final optimization model.

5.2.2 Do current policies internalize in an economically efficient way the externalities of soil pollution? Do current policies correctly internalize the externalities of phytoremediation?

5.2.2.1 Food- and soil policy

The preparatory study to the proposal for a European Soil Framework Directive raises that contaminated land management should follow the concept of risk based land management[120], based on a case by case approach. Moreover, it mentions that every Member State should work out a national action plan for contaminated land management but that an EU-wide inventory or a harmonization of plans on EU level is inappropriate because many Member States have a different concept of a contaminated site. This means in particular that soil remediation lies within the responsibility and interpretation of each Member State (Van Camp *et al.*, 2004). Indeed, the Soil Framework Directive proposal states the following: *"In order to successfully prevent and limit risk to human health and the environment stemming from soil contamination, Member States should identify the sites which according to their assessment are posing a significant risk in this regard."* However, at the same time, the Framework acknowledges that: *"It is recognized that different risk assessment methodologies for contaminated sites are currently being applied in Member States. In order to move towards a common approach ensuring neutral conditions of competition and a coherent soil protection regime, a thorough exchange of information is needed to establish the suitability of harmonizing some of the elements of risk assessment as well as to further develop and improve the methodologies on eco-toxicological risk assessment."* This actually means that in the final form of the Soil Framework Directive the definition of risk

[120]As defined by the CLARINET action (2002a): "*Risk describes the combination of the probability and the effects of contamination, for example adverse environmental effects on human health, on ecosystems, or on water resources. If an adverse effect has occurred, the consequences are often described as damage***."**

and what is conceived as unacceptable high risk might be harmonized between the different member countries, but that Member States can decide for themselves how they deal with this risk.

Provoost *et al.* (2006) identified in their review on soil cleanup standards in Europe that differences between countries are mostly due to scientific elements, political elements (such as the determination of "excess cancer risk"), differentiation in land use types, and the inclusion of background exposure (exposure not related to soil contamination). Therefore, the suggested thematic strategy for soil protection could indeed initiate the harmonization of the general model to increase transparency in model use, but adjustments for local parameters should still be necessary.

Control on application of legislation is inconsistent. Based on food threshold values, some harvests have been confiscated in the past, but the practice of strictly applying the law is not consistent. Moreover, crops from farmers in the region are subject to more thorough control, even though soil samples tested for soil standards might indicate that these farmers practice in a safe zone. Moreover, the system of control appears to be a trapped system, where a first check is based on soil standards, resulting in a safe and unsafe farming zone (classification made by authors). Soil samples above the accepted soil standard result in crops from unsafe zones being tested thoroughly regarding food threshold values. However, some food threshold values lie below background values for Cd, and while farmers growing crops in a safe zone come away with these crops, farmers from the unsafe zone see their harvests being confiscated, which is unfair. The trapped system of dividing soils up into two classes based on soil standards (BSN) (for pH = 5, below 2 mg kg^{-1} is safe, and above 2 mg kg^{-1} is unsafe) is not precise enough to cover all food threshold values. Several vegetables slip through the mazes. Following soil standards, the safe zone (< 2 mg kg^{-1}) allows all crops. This might indicate that soils should be divided into more classes. A third class might be introduced based on the background value (BV), resulting in a safe (below BV), relatively safe (below BSN), and unsafe (above BSN) class. Strictly interpreting current food threshold values, some crops shouldn't be grown on most Belgian (even uncontaminated sandy) soils.

This is already the case for some vegetables in the Campine region, but in an inconsistent way. It might however be more appropriate to reconsider the general BV of 0.7 mg kg^{-1} for all soils and make it specific for different soil textures. Doing this will lead to the exclusion of scorzonera and celery on Belgian sandy soils (as they require soil concentrations of Cd below 0.13 mg kg^{-1}).

In Belgium, food- and fodder threshold values, and soil standards exist simultaneously to evaluate risks of (Cd) contaminated agricultural soils. Both are based on a human toxicological risk assessment of contaminants. The basis for the food- and fodder threshold values and soil standards lacks transparency on the perception of risk. The cut-off value for the risk assessment, *i.e.* the maximum allowed risk, is based on the precautionary principle, providing "adequate" safety margins for human or ecological health. For European food- and fodder threshold values for Cd, the Commission Regulation on foodstuffs bases its values on an advice of 2 June 1995 where the Scientific Committee on Food (SCF) attaches its approval to a provisional tolerable weekly intake (PTWI) of Cd of 7 µg kg^{-1} body weight. In the SCF report, the committee concluded that it was impossible to exclude carcinogenic risk from dietary exposure to Cd and was therefore unable to set a safe level of allowed Cd in foodstuffs. As a result, 7 µg kg^{-1} body weight was chosen as a very careful weekly level to keep dietary exposure to Cd as low as possible.

When soil standards are determined based on a representative average food basket, the use of soil standards leads to the same results as the use of food- and fodder threshold values. From the moment soil standards and food threshold values do not lead to the same result and one approach should be preferred over another, there will be unnecessary harvests. The situation nowadays in the Campine region combines both soil standards and food threshold values to reduce transaction costs. However, it is not applied in a consistent way.

Contamination is concentric around the smelters. It could therefore also be expected that farmers in the region are facing different concentrations of Cd, with farmers in the near vicinity of the smelters facing highest Cd concentrations. Resultingly, the use of soil standards will benefit some farmers, whereas the use of food- and fodder threshold values will benefit others: the application of food- and fodder threshold values is beneficial for farmers when initial concentrations (C_0) lies between 10.8 and 1.5 mg kg^{-1} soil. Comparing different soil standards, we notice that this only results in a shift in R over the C_0: the exact same R is reached at a higher C_0, in case soil standards are less strict.

Soil standards and food- and fodder threshold values could lead to the same total welfare (depending on the distribution of agricultural land within the contaminated zone). The stricter the standard, the more contaminated land should be located in the less contaminated zone to lead to the same economic welfare. This means that soil standards, which are based on an average food basket, are determined at the approximately right level. However, no conclusion could be reached about this since this is accidental, since the model's decisions are not based on what is consumed but what is economically optimal to grow on the contaminated land.

We did not take into account control costs. As we might expect, these are higher for food threshold values than for soil standards. Hence we see in Flanders a current practice of applying soil standards before food threshold values. Soil standards only serve as a first criterion to determine whether food threshold values will be checked. An argument for the introduction of food threshold values after soil standards (and not just soil standards) is the fact that one could argue that from the moment crops are harvested soil standards no longer apply. This would then imply that first soil standards should be applied. When crops are harvested they should then comply with food- and fodder threshold values.

Standards could reduce soil pollution in a cost-efficient and -effective way if they were chosen precisely at the level of pollution where marginal abatement cost (MAC) curves of remediation equal marginal damage cost (MDC) curves. However, such standards impose a high information- and control burden on authorities, leading to a (rather inconsistent) follow-up. Regulations tend to stop the development of new pollution controlling technologies because there just are no economic incentives for agents to exceed control targets. In our example, the economically efficient soil standard, with a MAC based on conventional remediation costs would be very high. This is because conventional remediation technologies (BATNEEC) are very expensive. This high threshold value shows that conventional remediation technologies are actually not viable on moderately contaminated sites: risk for human health containment cannot be reached within reasonable costs, leading to soil standards that lie high above current soil standards. Alternatively, plant-based technology costs could be used to determine MAC. MAC of phytoremediation is equal to the loss in income due to phytoremediation and depends on i (4%), the number of years of remediation, the income of the remediation crop, the income after remediation, and the cost of conventional remediation. Since we assume linear accumulation during phytoremediation, and a 100% effectiveness, the physical effectiveness is constant. If plant-based technologies would be supported as the best available technology, it could be economically efficient to continue remediation until Cd in the soil reaches very low levels.

5.2.2.2 Energy policy

In August 2009, the United States Environmental Protection Agency published its proposal called Superfund Green Remediation Strategy which outlines strategic recommendations for cleaner site redevelopment (EPA, 2010). Green remediation factors might even be included in the evaluation of the economic efficiency. When growing crops for energy production on land, while gradually remediating the soil, a policy suggesting government intervention based on CO_2 abatement might be necessary to improve economic efficiency. This would only be allowed if the external benefit of CO_2 abatement is not included correctly in the price of biomass and as such not yet taken into account in economic optimization. By subsidizing renewable energy based on its external benefit (*i.e.* its CO_2 abatement and valuation) the economics on which the farmer will base its crop decision will be altered the correct way. Taxes and subsidies are economic incentive instruments to achieve a given level of environmental quality at least cost, and encourage behavior through price signals rather than through specific target levels of pollution (Hahn and Stavins, 1992). The marginal damage function of CO_2 is a flat function and under- or overestimating MAC CO_2 when MDC is flat creates no dead weight loss in a tax- or subsidy-system, as opposed to a regulatory system. As opposed to a tax system, the subsidy system requires money from the regulator and knowledge of the initial level of pollution. Therefore, we conclude that taxing "bad stuff" is preferred over subsidizing "good stuff".

Currently, the external effect of CO_2 abatement is not internalized correctly in the biomass price and this might have consequences, not only on phytoremediation as an alternative plant-based technology for conventional technologies, but also on the choice between alternative crops. Our analysis shows that, based on CO_2 abatement, current Flemish subsidies do not succeed in sending the right price signals for biomass based renewable energy production. Certificates in Flanders provide financial stimuli for sources in function of environmentally friendly or emission poor fuels but (i) do not guarantee an automatic and full inclusion of external costs of fossil energy or external benefits of green energy and (ii) are not neutral for different technologies concerning the technology's potential positive impact on the environment. The subsidies studied here are much higher than could be justified by CO_2 abatement alone, except for the case of separate heat production. Implications for the plant-based technology are not straightforward. The corrections might generate internal shifts amongst alternative crops as one crop might gain advantage over another crop. Although it might be true that conversion installations are not capable of offering biomass prices completely corrected for CO_2 abatement, this would be true for uncontaminated as well as for metal enriched biomass. Moreover, including the correct price for CO_2 to fossil prices could make biomass more competitive as an energy fuel input and thus offer opportunities for plant-based technologies. Finally, due to the raise in biomass prices, phytoremediation would not lose its competitive advantage over conventional technologies although this will have to be studied over a full phytoremediation management life cycle.

5.2.2.3 Metal waste disposal policy

If biomass after phytoremediation would be treated as if there were no metals present, the metals would eventually end up somewhere else without control (through the smoke stack or application of the rest products on the field, …) and the whole purpose of phytoremediation would just have been to generate an income, combined with renewable energy for the farmer, while at the same time generating costs for other stakeholders. Therefore, it is important that biomass and rest products are properly used and disposed off. Concerning waste disposal, the relevant legislation is the Council Directive 1999/31/EC on the landfill of waste. This Directive raises that Member States should take measures to ensure that all costs involved in the set-up and operation of a landfill site, as well as the costs to ensure that obligations including after-care provisions are covered, should be included in the price of waste disposal. The activities and requirements to which these costs refer are mentioned in the Annexes of this Directive. Annex I contains general requirements that should be met by all landfills to ensure that the landfill does not pose a serious environmental risk. Annex II then further defines that criteria for acceptance at a specific class of landfill must be derived from considerations pertaining to protection of the surrounding environment, protection of the environmental protection systems, protection of the desired waste-stabilization processes within the landfill, and protection against human health hazards. Annex III finally defines the control- and monitoring procedures in operation and after care phases. Based on this we decided for now that current policies concerning secondary use, landfilling, … correctly internalize the effect of metals on human health and the environment. They therefore lead to the economically most efficient solution for metal enriched biomass and rest products.

5.2.3 Does phytoremediation offer an economically viable alternative for conventional remediation technologies and which crops should be used and under what circumstances?

The optimization model builds on food- and fodder threshold values. The BeNeKempen project defined for different Cd concentrations in soil and for different pH levels, the probability of exceeding legal threshold values for food and fodder for 16 agricultural crops, maize, and grass (HI crops). Based on this, the maximum Cd concentration in soil (C_q) for which the crop can safely be grown without exceeding Cd threshold values for food and fodder use in 50%, 10%, and 5% of measurements could also be calculated. Based on accumulation capacities of each of the remediation crops, we could then calculate how long it would take to reach C_q, starting from an initial level of contamination C_0.

The model reduces the decision for the combination of one remediation crop with one HI crop to a one time decision in year 0. The model is based on the adapted gross income (AGI), a method of measurement specific for our purpose. AGI_{HI} is based on the gross balance and does not include wages, building costs and selling (auction, promotion) costs. However, we did add equipment, third party labor, and fuel costs. AGI_{rem} represents yearly income during remediation from time 0 until time x. AGI_{HI} represents yearly income after remediation from time x until ∞. We took into account two additional aspects: HI crops cannot be grown year after year, and good agricultural practice respects a certain order in growing different crop families. Adjustments for rotation schemes result in AGI'. For each remediation crop, AGI'_{rem} is corrected for external costs and benefits of renewable energy production and metal waste disposal (through current policy/legislation), resulting in r_{rem}. This r_{rem} is then discounted at a social discount rate of 4% and over time x, resulting in R_{rem}.

In the next step the model maximizes $R = R_{rem} + R_{HI}$ for each C_0. HI crops are grown from time x+1 until infinity, *i.e.* over ∞-x. To add R_{HI} to R_{rem}, the former is first discounted from year x to year 0. Finally, the maximizing step consists of two phases. First, for every C_0-C_x combination, the model decides whether energy maize, rapeseed, or willow will lead to the highest R. Second, the model calculates for every C_0 the C_x which would lead to the highest R. The outcome of the optimization model is the combination of one remediation crop and one HI crop for every initial level of contamination (C_0), resulting in an optimal end level of contamination (C_x), maximizing net benefit (R, € ha^{-1} over infinity, and at an interest rate of 4%). The outcome depends on trade-offs: (i) between higher income during remediation and faster remediation, and (ii) between different HI crops.

From the model we learn that, in general, energy maize could be grown as an alternative and safe crop for ranges of Cd concentration in soil between 12 and 10.1 mg kg^{-1}, after which maize for grain (!) could be grown (maximum allowed soil concentration is 10 mg kg^{-1}). Willow is chosen between 3.9 and 3.5 mg kg^{-1} until 3.4 mg kg^{-1}, after which asparagus can be grown, and from 0.6 mg kg^{-1} to remediate until 0.5 mg kg^{-1} from which moment endive is grown. For a HI crop to be chosen by the model, the HI crop should represent a good combination of high AGI and easy to reach. In the base case asparagus (AGI = € 3,619 and C_q = 3.4) is preferred over beans (AGI = € 2,616 and C_q = 2.7). However, when pH is raised (*e.g.* by liming) to 5.5 (in the base case the pH = 5), metals become less available and C_q for all HI crops raise, resulting in beans to be preferred over asparagus under certain conditions.

Rapeseed is in competition with energy maize, and our analyses show that energy maize and rapeseed are equally preferred when $AGI_{RS} = r_{EM}$. Energy maize is the preferred crop for large distances to target (DTT), *i.e.* the difference between C_q and C_0, while willow is preferred for average to small DTT. Changing the price of willow (*ceteris paribus*), or changing the costs of willow (*ceteris paribus*) has no effect.

Given the contamination range (12-0 mg kg^{-1}), the model only suggests first growing one of the alternative crops in 15% of cases. In all other cases, the model suggests immediately growing the allowed crop. A raise in biomass yield of willow results in willow to be chosen more often as the remediation crop, and more often actual remediation is suggested by the model. To induce actual remediation (50% of cases), the AGI of willow should at least be € 1,200 ha^{-1} $year^{-1}$. Our analyses show that this could not be reached by changing parameters separately, but a simultaneous increase in allowed rotation cycles, an increase in biomass yield, a reduction in costs, and the harvest of leaves might be able to realize this.

These results are coming from the model when it is based on food- and fodder threshold values. However, soil standards exist simultaneously. Given C_0 > 2 mg kg^{-1}, which is the BSN above which soil should be remediated according to soil standards, the model never suggests to remediate soil until C_x < 2 mg kg^{-1}, simply because this is not the economically optimal solution. Since soil standards are much lower than in the case of food- and threshold values, remediation will at any case take very long. This is a disadvantage for willow, as this crop benefits from short distances to target (DTT, *i.e.* C_x – C_0). As a result, energy maize is most often preferred as the alternative crop in the case of soil standards. It is a choice for sustainable land use and management instead of remediation as such. Intuitively, energy maize would be preferred by farmers since for them it is a familiar crop not requiring big adjustments in their equipment. Therefore, it will be more easily accepted by farmers. We suggest growing energy maize to aim at risk reduction, to generate an alternative income for farmers, and in the very long run also to realize a gradual reduction of the pollution levels. We zoomed in on this crop and found that the effect on the income per hectare of growing energy maize instead of fodder maize should be positive. The probability on a positive extra income is 90% when the farmer grows energy maize instead of fodder maize and simultaneously exploits the digester in cooperation with other farmers.

One of the drawbacks of the optimization model lies in its practical applicability. The model optimizes for each Cd concentration in soil (C_0) the income over infinity. The infinite time line does not appeal to current farmers. Moreover, a different approach for each of the hectares does not take into account economies of scale. Also, the model might be hard to understand. Taking into account the previous comments on the acceptability of crops, a second model was developed that started with a given number of hectares of willow combined with traditional and easily acceptable crops. This model then calculates maximum DTT, and within these DTT, maximizes R per DTT (instead of per C_0), within a limited time period of 40 years.

The second model confirms that phytoremediation works if only physical data are considered. If these physical data are linked to economic data, results become somewhat more complicated. A first important conclusion is that phytoremediation is in any case more cost-effective than conventional remediation, within limited contamination ranges. Given one hectare of land with a DTT of 1.05 mg kg^{-1} (since this is the maximum DTT the phytoremediation technology can cover and makes comparison with conventional remediation possible): should one best grow SRC of willow (most effective option) or apply conventional remediation? Conventional remediation cannot compete with SRC of willow, unless the cost of conventional remediation goes down drastically, or the AGI of HI crops raises substantially. The purpose of the second model is to reduce this cost of SRC based phytoremediation by including it in a crop model. By including other alternative crops such as energy maize with a much higher AGI, the average cost of remediation per remediated ha goes down drastically, depending on the crop scheme. A second important conclusion from this study is that SRC of willow shows the physical and economic (if combined with HI crops) potential to sustainably remediate moderately Cd enriched land within a period of 40 years, with calculations even showing increases in NPV(AGI) at low levels of contamination, if and only if the optimal cropping scheme as suggested by the model is employed, and within contamination ranges defined by the model. A third conclusion that can be drawn from the analysis is that, based on this model, remediation will at any case cost money. Until now no initiative had been taken to actually calculate the cost per hectare and it was and still is generally assumed that these costs should and will be borne by the farmer, with the resulting lack of action by stakeholders and the tolerance thereof by local and federal authorities. Therefore, we developed a suggestion for the Flemish government to bear remediation costs: depending on the height of her support for the involved farmers, she decides how fast soils will be remediated, *i.e.* based on a goal based support mechanism. We calculated a level of support (€ ha^{-1} $year^{-1}$) over 40 years that would encourage local farmers to grow the suggested rotation schemes that contribute to the gradual remediation of the Cd enriched soil.

The model demonstrates that phytoremediation could be used on moderately contaminated land and has a competitive advantage over conventional remediation. Moreover, it shows that with the correct incentive, government can guide the intensity of remediation. However, what the model does not do is convince the tenant to remediate, since we assume that the tenant (in this case the farmer) is still allowed to grow fodder maize on the land. The responsibility to convince land owners to remediate land lies with policy makers, who are greatly depending on studies which demonstrate total benefits (*i.e.* private and social) of remediation. Given current BSN, current food threshold values below which HI crops can be grown safely, and small DTT's that can currently be covered by remediation crops, the model ends up remediating soils with a concentration that lie below BSN.

5.2.4 Does phytoremediation offer a multifunctional and sustainable alternative for conventional remediation technologies for functional repair or management of metal contaminated agricultural sandy soil?

The concept of sustainability consists of 2 components: intergenerational equity and dynamic efficiency. Intergenerational equity on the one hand emphasizes the ability of the economy to maintain living standards, *i.e.* intertemporal social welfare (V_t) should not decrease over time. Dynamic efficiency on the other hand focuses on maximizing V_t (at every time period t), *i.e.* the integral of discounted values of current and future utility from society's aggregate consumption from time t to infinity.

Does the use of plant-based technologies for soil management of farm land contribute to intergenerational equity? Potentially yes. Privately, the farmer will be able to sustain his income by growing biomass for remediation- and energy purposes. Moreover, socially, by producing energy from the biomass, we do not only make it possible for future generations to produce on the previously contaminated land, but also safeguard the energy potential for future generations. Also, performing research on the promising technology of phytoremediation nowadays (optimizing accumulation potential, treatment of the biomass, conversion techniques, …) contributes to the concept of sustainable development, as we provide the future with a ready to use technology which they will on their turn use for sustainable land management.

Does the use of plant-based technologies for soil management result in dynamic efficiency? Potentially yes. The model searches for the economically efficient combination of remediation crop with conversion technology and high income crop. This maximizes social welfare at each time t, given that the suggested pathway is followed, given our limited analysis based on a private economic analysis completed with the renewable energy potential and safe metal treatment, and given that policies (on which the model is based) are correctly internalizing external effects.

In literature one can often find the term multifunctional agriculture (see *e.g.* Dent *et al.*, 1995; OECD, 2001). Multifunctionality refers to the fact that an economic activity can have multiple outputs and, by consequence, may contribute to several societal objectives at once (OECD, 2001). This can become policy relevant if, among the multiple outputs generated, there are some that are welfare-enhancing or welfare-reducing but for which no private markets exist. If, in such a case, a policy is deemed necessary to meet the demands of society, to internalize an externality, the characteristics of the activity involved have implications for the design of the correcting policy. Current environmental policies try to withhold us from burdening the environment, from free-riding on the future (Solow, 1991). There is a dual connection between environmental policies and sustainability. On the one hand, the environment needs protection by public policy because by damaging the environment we can profit and at the same time have (some of) the costs borne by others, by the future. On the other hand, sustainability is a problem precisely because its definition implies and we all know that we can make profits at the expense of the future (free-riding) (Solow, 1991). Of course we will make mistakes in our calculations, and in our suggestions for policy development on sustainable land use and management. However, this should not refrain us from moving forward. When we consistently make mistakes in under- or overestimating marginal benefits and costs, we can at least say whether a policy is increasing social welfare and demand for its implication.

Chapter 5.3 *Future research*

Total metal removal rate and resulting remediation duration depend on soil characteristics, level of metal contamination, level of available metals, metal extraction by the plant, and biomass production of the plant (Koopmans *et al.*, 2008; Vangronsveld *et al.,* 2009). As metals are removed by the plants, not just the total metal contamination reduces, but also the concentration of available metals reduces. This relation is in most cases logarithmic, meaning that the concentration of available metals reduces faster than the level of total metals in the soil, reaching a limit level of available metals. When long-term field experiments for the sandy Campine soils become available, this deserves more attention.

Another potential crop for remediation of Cd contaminated soils is tobacco (*Nicotiana tabacum*). In several studies it demonstrates a great ability to accumulate Cd, while maintaining its high biomass production (Vangronsveld *et al.*, 2009). Moreover, accumulation is mainly concentrated in above-ground parts (Mench *et al.*, 1989; Dorlhac de Borne *et al.*, 1998), which is a necessary trait for effective extraction. Liang *et al.* (2009) and Vangronsveld *et al.* (2009) compared the ability to remediate a moderately Cd contaminated soil of tobacco with several (hyper)accumulators. The former authors used published data to estimate the phytoextraction potential, while the latter authors conducted a series of field experiments. Both studies concluded that tobacco would reach the target concentration in less time than the hyperaccumulators. However, due to the high concentration of Cd, harvested tobacco biomass from phytoextraction can no longer be used for smoking purposes. Therefore, alternative uses have to be found. Currently, options are rather limited. Meher *et al.* (1995) explored the possibilities of energy production from tobacco and found that tobacco waste was amenable to anaerobic digestion. Also, biorefineries could convert the tobacco biomass to valuable products for industrial markets (Kamm and Kamm, 2004). Following the latter concept, Machado *et al.* (2010) recovered solanesol from tobacco, an alcohol employed by the pharmaceutical industry, and a valuable by-product. These studies show that tobacco is a potential phytoremediation crop which merits further research. This research should then focus on exploring economic opportunities of (contaminated) tobacco biomass for alternative applications.

Crops used for plant-based management of trace element enriched soils need alternative application. In this study we have chosen for energy conversion as a sustainable alternative because energy production will more likely get public approval. In Europe, farmers are variously rewarded for direct positive contributions to biological diversity (particularly wildlife habitat), improvements (or avoided negative impacts) to water quality and increased soil health through the concept of cross compliance in the European Common Agricultural Policy (CAP). Many countries also support bioenergy programs, with the intent to promote the production of cleaner fuels instead of fossil fuel. Moreover, on a global scale, the increasing interest in carbon sequestering effects of many types of agriculture points to a growing number of programs in the near future that will support certain farming practices as a way of improving overall air quality (DeVries, 2000). Moreover, there has already been research on tracing metals in energy conversion installations. Until now, there had been no research yet on tracing metals in other biomass using technologies (*such as* paper mills). However, this is about to change. The numerous phytoremediation research projects that Hasselt University is working on with several other universities has caught the attention of (amongst others) Sappi Lanaken. This is a local paper mill who is interested in investigating the possibilities of using poplar from contaminated fields for paper production (SAPPI, personal communication, August 2010). This requires wood stems with diameters of minimum 7 cm. Therefore, 10 are of 3 poplar clones have been planted on the experimental field to investigate whether it would be possible to reach this diameter within a cycle of 8 years, within which the biomass has to be harvested to be considered as an agricultural crop and not as forest.

Moreover, the external costs we calculated are only a subset of total external costs. According to Hall and Scrase (1998) caution is needed to ensure that other issues are not considered unimportant simply because they are not expressed in monetary terms. We should acknowledge these limitations when coming to policy oriented recommendations based on external cost data (Krewitt, 2002). In literature, other potential externalities of soil remediation relate to biodiversity issues, water control, vegetation filters, biological activity in the soil, carbon sequestration, waste handling, and erosion control (Burger *et al.*, 2004; Licht and Isebrands, 2005; Mirck *et al.*, 2005; Zalesny *et al.*, 2009). The study performed by Van Wezel *et al.* (2007) gives a comprehensive overview of other benefits of less contamination in soil such as positive health effects, improved drinking water, increased property values. Not all of them are technology specific, *i.e.* could contribute to the competitive advantage of plant-based remediation over conventional remediation. In this study we handled 2 externalities, but the extended technical literature on externalities demands for an economic assessment of them.

Groundwater is an essential resource that sustains mankind and the various ecosystems that depend on it. Therefore, it is important to manage and protect it properly. If groundwater contamination occurs, it is important to remediate or at least contain the contamination in order to prevent it from reaching lakes and rivers by natural discharge (UN, 2006). Phytoremediation has proven its effectiveness for the remediation of contaminated groundwater. Due to their high evapotranspiration rates, rapid growth, and phreatophytic root development, poplar trees are considered as ideal candidates to remediate groundwater (Barac *et al.*, 2009; Weyens *et al.*, 2009c). They are perceived as inexpensive solar powered pumps that can be used to install hydraulic barriers (Schnoor *et al.*, 1995; Al-Yousfi *et al.*, 2000). Even though phytoremediation is reported being a cost-effective remediation technology, until now only a limited number of phytoremediation projects were implemented. EPA (2010) reports 17 sites at which phytoremediation is applied for groundwater remediation, which is 5% of all *in situ* groundwater treatment projects. In the European Union, the use of phytotechnologies is also limited (Vangronsveld *et al.*, 2009).

It is very hard for new technologies to enter the market. This explains the fact that conventional remediation is still preferred in almost 100% of remediation projects (natural attenuation not taken into account) over green remediation technologies such as phytoremediation (Public Waste Agency Flanders, OVAM, personal communication, January 2011). Given the fact that in a perfect market government intervention would not improve social welfare, subsidies and taxes are only allowed to correct for market imperfections, such as externalities. Costello and Finell (1998) point out that regulatory factors can create technology development opportunities that would not exist in an economic system without any government intervention, *i.e.* government intervention should be allowed if it is intended to stimulate new technologies. However, this raises many issues:

from what moment do we consider a new technology market ready, how far does this stimulation go, and when do we stop stimulating? Moreover, internalization of externalities in biomass prices is a nice approach in theory. How this would be translated and monitored practically is not clear yet, since this would result in different biomass prices depending on its end use. This should not withhold us from studying externalities of phytoremediation since it is these findings that support its label as a sustainable technology, which will only stimulate other research and thus the introduction of this technology on a commercial scale.

LITERATURE

Abrahamson, L. P., Volk, T. A., Kopp, R. F., White, E. H., Ballard, J. L. (2002) Willow biomass producer's handbook. State University of New York, Syracuse

Actief Bodembeheer de Kempen (ABdK) (2008) Cadmium op landbouwpercelen in de Kempen. Teeltadviezen voor landbouwgewassen. ABdK, Eindhoven. Available from www.abdk.nl

Adegbidi, H.G., Volk, T.A., White, E.H., Abrahamson, L.P., Briggs, R.D., Bickelhaupt, D.H. (2001) Biomass and nutrient removal by willow clones in experimental bioenergy plantations in New York State. Biomass Bioenerg 20, p. 399-411

Adriaensen, K., Bert, V., Bohm, K., Brignon, J-M., Cochet, N., Cundy, A., Denys, S., Friesl-Hanl, W., Gombert, D., Haag, R., Hurst, S., Jaunatre, R., Jollivet, P., Kumpiene, J., Magnie, M-C., Marschner, B. ,Mench, M., Mikhalovsky, S., Muller, I., Onwubuya, K., Puschenreiter, M., Raspail, F., Renella, G., Rouil, L., Ruttens, A., Schoefs, O., Soularue, J-P., Stolz, R., Tack, K., Teasdale, P., Tlustoš, P., Vangronsveld, J., Vialletelle, F., Waite, S. (2008) Sustainable management of trace element contaminated soils (SUMATECS)— development of a decision tool system and its evaluation for practical application. Project no. SN-01/20. Final Research Report. Universitat fur Bodenkultur Wien (BOKU), Vienna. Available from www.snowman-era.net/downloads/SUMATECS_FINAL_REPORT.pdf

Aidt, T.S. (1998) Political internalization of economic externalities and environmental policy. J. Public Econ 69, p. 1-16

Al-Yousfi, A. B., Chapin, R. J., King, T. A., Shah, S. I. (2000) Phytoremediation—The Natural Pump-and-Treat and Hydraulic Barrier System. Pract. Period. Hazard., Toxic, Radioact. Waste Manage. 4(2), p. 73-77

Alkorta, I., Hernandez-Allica, J., Becerril, J.M., Amezaga, I., Albizu, I., Garbisu, C. (2004) Recent findings on the phytoremediation of soils contaminated with environmentally toxic heavy metals and metalloids such as zinc, cadmium, lead and arsenic. Rev. Environ. Sci. Biotechnol. 3, p. 71-90

Ananda, J., Herath, G. (2008) Multi-attribute preference modelling and regional land-use planning. Ecol. Econ. 65, p. 325-335

Andreoli, M., Tellarini, V. (2000) Farm sustainability evaluation: methodology and practice. Agric., Ecosyst. Environ. 77, p. 43-52

Animal Sciences Group (2006) Handboek snijmaïs. Animal Sciences Group, Lelystad, The Netherlands

Arrow, K., Cropper, M.L., Eads, G.C., Hahn, R.W., Lave, L.B., Noil, R.G., Portney; P.R., Russell, M., Schmalensee, R., Smith, V.K., Stavins, R.N. (1996) Is there a role for Benefit-Cost Analysis in Environmental, Health and Safety regulation? Sci. 272, p. 221-222

Arrow, K., Dasgupta, P., Goulder, L., Daily, G., Ehrlich, P., Heal, G., Levin, S., Maler, K-G., Schneider, S., Starrett, D., Walker, B. (2003) Are we consuming too much? J. Econ. Perspect., 42 p.

Assunçao, AGL., Schat, H., Aarts, MGM. (2003) *Thlaspi caerulescens*, an attractive model species to study heavy metal hyperaccumulation in plants. New Phytol. 159, p. 351–360

Azar, C., Sterner, T. (1996) Discounting and distributional considerations in the context of global warming. Ecol. Econ. 19, p. 169-184

Barac, T., Weyens, N., Oeyen, L., Taghavi, S., van der Lelie, D., Dubin, D., Spliet, M., Vangronsveld, J. (2009) Field note: hydraulic containment of a BTEX plume using poplar trees. Int. J. Phytorem. 11(5), p. 416-424

Barde, J-P. (2000) Environmental policy and policy instruments. In: H. Folmer, H.L. Gabel (Eds.). Principles of environmental and resource economics. Edward Elgar, Northampton, UK, p. 157-201

Bardos, P., Chapman, T., Andersson-Skold, Y., Blom, S., Keuning, S., Polland, M., Track, T. (2008) Biomass production on marginal land. BioCycle International December 2008, p. 50-52

Baxter, L. (2005) Biomass-coal co-combustion: opportunity for affordable renewable energy. Fuel 84, p. 1295–1302

Baumol, W.J., Oates, W.E. (1988) The theory of environmental policy. Second edition. Cambridge University press, New York, US

Bennett, FA., Tyler, EK., Brooks, RR., Gregg, PEH., Stewart, RB. (1998) Fertilisation of hyperaccumulators to enhance their potential for phytoremediation and phytomining. In: Brooks R. (ed) Plants that hyperaccumulate heavy metals. CAB International, Wallingford, UK, p. 249–259

Berndes, G., Fredrikson, F., Borjesson, P. (2004) Cadmium accumulation and Salix-based phytoextraction on arable land in Sweden. Agr. Ecosyst. Environ. 103, p. 207-223

Berndes, G., Borjesson, P., Ostwald, M., Palm, M. (2008) Multifunctional biomass production systems-an overview with presentation of specific applications in India and Sweden. Biofuels, Bioprod., Bioref. 2, p. 16-25

Bernhard, R., Verkleij, J.A.C., Nelissen, H.J.M., Vink, J.P.M. (2005) Plant-specific responses to zinc contamination in a semi-field lysimeter and on hydroponics. Environ. Pollut. 138, p. 100-108

Bickel, P., Friedrich, R. (2005) Externalities of Energy, Methodology 2005 Update. Office for official publications of the European Communities, Luxembourg

Bierkens, J., Cornelis, C., De Raeymaecker, B., Schoeters, G., Hooghe, R., Verbeiren, S., Nouwen, J. Provoost, J., Waegeneers, N., Oorts, K., Smolders, E., Ruttens, A., Vangronsveld, J., Van Geert, K., Schoeters, I., Ghyoot, W., Van Gestel, G., Geysen, D., Dedecker, D. (2009) Aanvulling bij basisinformatie voor risico-evaluaties. Zware metalen en arseen: stofdata. OVAM, Mechelen

Bierkens, J., De Raeymaecker, B.,Cornelis,C., Schoeters, G., Hooghe, R., Verbeiren, S. Ruttens, A., Vangronsveld, J., Smolders, E., Schoeters, I., Van Geert, K., Van Gestel, G., Geysen, D., Dedecker, D., Van De Wiele, K. (2010) Voorstel voor herziening bodemsaneringsnormen voor cadmium. OVAM, Mechelen

Boardman, A.E., Greenberg, D.H., Vining, A.R., Weimer, D.L. (2006) Cost-Benefit Analysis. Concepts and Practice. Third Edition. Pearson, Prentice Hall, New Jersey

Borjesson, P. (1996) Energy analysis of biomass production and transportation. Biomass Bioenerg. 11(4), p. 305-318

Borjesson, P. (1999a) Environmental effects of energy crop cultivation in Sweden-I: Identification and quantification. Biomass Bioenerg 16, p. 137-154

Borjesson, P. (1999b) Environmental effects of energy crop cultivation in Sweden-II: Economic Valuation. Biomass Bioenerg 16, p. 155-170

Bom, J. (n.d.) Aardgascentrale minder CO_2 – uitstoot dan kerncentrale. Öko – Institut, Germany. Available from
http://www.peopleplanetprofit.be/artikel.php?IK=1053

Bowen, A. (2007) Comments by Alex Bowen on DEFRA draft dated 10/08/07

Braschkat, J., Gartner, S.O., Reinhardt, G.A., Uihlein, A. (2003) Biogas versus other biofuels: a comparative environmental assessment. 2-4 oktober 2003, Esbjerg

Brent, R.J. (2003) Cost-benefit analysis and health care evaluation. Edward Elgar Publishing limited, Cheltenham, UK

Burger, J., Carletta, M.A., Lowrie, K., Miller, K.T., Greenberg, M. (2004) Assessing ecological resources for remediation and future land uses on contaminated lands. Environ. Manage. 34(1), p. 1-10

Calus, A., Dumoulin, A., Ghekiere, G., Mattheeuws, B., Platteau W. (2007) Vergisting op boerderijschaal: techniek, grondstoffen en eindproducten. Available from
http://www.vcm-mestverwerking.be/publicationfiles/brochurebiogas2007.pdf.

Castle, E.N. (1970) The market mechanism, externalities, and land economics. J. Farm Econ. 47(3), p. 542-556

Cenni, R., Janisch, B., Spliethoff, H., Hein, K.R.G. (2001) Legislative and environmental issues on the use of ash from coal and municipal sewage sludge co-firing as construction material. Waste Manage. 21, p. 17-31

Ceulemans, R., McDonald, A.J.S., Pereira, J.S. (1996) A comparison among eucalypt, poplar, and willow characteristics with particular reference to a coppice, growth-modelling approach. Biomass Bioenerg. 11(2/3), p. 215-231

Chaney, R.L. (1983) Plant uptake of inorganic waste constituents. In: Parr, J.F., Marsh, P.B., Kla, J.M. (Eds.). Land Treatment of Hazardous Waste. Noyes Data Corporation, Parkridge, USA, p. 50-76

Chaney, R.L., Li, Y.M., Brown, S.L., Angle, J.S., Baker, A.J.M. (1995) Hyperaccumulator based phytoremediation of metal-rich soils. In: Current topics in plant biochemistry, physiology and molecular biology. Will plants have a role in bioremediation? Proceedings of 14th Annual Symposium. April 10–12, 1995. Columbia, Missouri, p. 33–34

Chaney, R.L., Malik, M., Li, Y.M., Brown, S.L., Brewer, E.P., Angle, J.S., Baker, A.J.M. (1997) Phytoremediation of soil metals. Curr. Opin. biotechnol. 8, p. 279-284

Chaney, R.L., Li, Y.M., Angle, J., Baker, A., Reeves, R., Brown, S., Homer, F., Malik, M., Chin, M. (1999) Improving metal-hyperaccumulators wild plants to develop commercial phytoextraction systems: approaches and progress. In: Terry, N., Banuelos, G. (eds). Phytoremediation of contaminated soil and water. CRC, Boca Raton, FL

Chaney, R.L., Reeves, P.G., Ryan, J.A., Simmons, R.W., Welch R.M., Angle, J.S. (2004) An improved understanding of soil Cd risk to humans and low cost methods to phytoextract Cd from contaminated soils to prevent soil Cd risks. BioMetals 17, p. 549–553

Chen, Y., Cheng, J.J., Creamer, K.S. (2008) Inhibition of anaerobic digestion process: A review. Bioresour. Technol. 99, p. 44-64

Cho, J.H. (1971) Externalities and land economics. Land Econ. 47(1), p. 65-72

Chrysafopoulou, E., Kadukova, J., Kalogerakis, N. (2005) A whole plant mathematical model for the phytoextraction of lead (Pb) by maize. Environ. Int. 31, p. 255-262

Cidad, V.G., Mathijs, E., Nevens, F., Reheul, D. (2003) Energiegewassen in de Vlaamse landbouwsector. Steunpunt Duurzame Landbouw, Leuven

Claeys, B. (2009) Steenkool kan niet zonder groene certificaten. Bond Beter Leefmilieu, Belgium

Clarinet (2002a) Sustainable Management of Contaminated Land: An Overview. Report prepared by the Concerted Action "Contaminated Land Rehabilitation Network for Environmental Technologies" (CLARINET), funded by the European

Commission, DG Research, under the Environment and Climate Programme and coordinated by the Austrian Federal Environment Agency

Clarinet (2002b) Remediation of Contaminated Land Technology Implementation in Europe. Prepared by Working Group "Remediation Technologies" of the Concerted Action "Contaminated Land Rehabilitation Network for Environmental Technologies" (CLARINET), funded by the European Commission, DG Research, under the Environment and Climate Programme and coordinated by the Austrian Federal Environment Agency

Clarkson, R., Deyes, K. (2002) Estimating the social cost of carbon emissions. Government Economic Service Working Paper 140. DEFRA, London

Clauw, F. (2007) The effect of soil contamination on real estate value: a hedonic pricing approach based on different risk measures. PhD dissertation. Hasselt University, Belgium

Coase, H. (1960) The problem of social cost. J. Law Econ., 23 p.

Cogen Vlaanderen vzw (2006) Basishandboek warmtekrachtkoppeling. Cogen, Belgium

Costello, R., Finell, J. (1998) Institutional opportunities and constraints to biomass development. Biomass Bioenerg. 15(3), p. 201-204

Crettaz, P., Pennington, D., Rhomberg, L., Brand, K;, Jolliet, O. (2002) Assessing Human Health Response in Life Cycle Assessment Using ED10s and DALYs: Part 1—Cancer Effects. Risk Anal. 22(5), p. 931-946

Crookes, D., de Wit, M. (2002) Environmental economic valuation and its application in environmental assessment: an evaluation of the status quo with reference to South Africa. Impact Assess. Proj. Appr. 20(2), p. 127-134

Dalgaard, T., Halberg, N. Porter, J.R. (2001) A model for fossil energy use in Danish agriculture used to compare organic and conventional farming. Agric., Ecosyst. Environ. 87, p. 51–65

De Becker, R., D'Hooghe, J., Mortier, P. (2009) Vlaamse bruto standaardsaldi voor de gewassen en de veehouderij (Periode 2000-2005). Departement landbouw en visserij, afdeling monitoring en studie, Brussel

De Boer, H.C., van Duinkerken, G., Philipsen, A.P., van Schooten, H.A. (2003) Praktijkrapport Rundvee 27, Alternatieve voedergewassen. Animal Sciences Group, Lelystad

De Temmerman, L.O., Hoenig, M., Scokart, P.O. (1984) Determination of "normal" levels and upper limit values of trace elements in soils. Z. Pflanzenernachr. Bodenk. 147, p. 687-694

De Temmerman, L., Vanongeval, L., Boon, W., Hoenig, M., Geypens, M. (2003) Heavy metal content of arable soils in northern Belgium. Water, Air Soil Pollut. 148, p. 61-76

De Turck, K., De Sloovere, A. De Naeyer, F., Van Dyck, E. (2009) Regulatory Initiatives and Strategies for Managing the Soil and Groundwater Contamination with Heavy Metals in the Campine Area, Belgium, Contaminated site management in Europe, 27-29 October 2009, Ghent, Belgium

Dejonckheere, W., Steurbaut, W., Drieghe, S., Verstraeten, R., Braeckman, H. (1996) Pesticide residue concentrations in the Belgian Total diet, 1991-1993. J. AOAC International 79(2), p. 520-528

Demirbas, A. (2003) Sustainable cofiring of biomass with coal. Energy Conv. Manage. 44, p. 1465–1479

Dent, J.B., Edwards-Jones, G., Mc-Gregor, M.J. (1995) Simulation of Ecological, Social and Economic Factors in Agricultural Systems. Agric. Syst. 49, p. 347-351

Deuninck J. (2008) Analyse van de rechtstreekse steun uit Pijler I aan de landbouwer: campagne 2006. Beleidsdomein Landbouw en Visserij, Afdeling Monitoring en Studie, Brussel

DeVries, B. (2000) Multifunctional Agriculture in the international context: a review. Available from
http://www.landstewardshipproject.org/mba/MFAReview.pdf

D'Hooghe, J., Campens, V. (2009) Vlaamse standaardopbrengsten voor de gewassen en de veehouderij (2003-2005). Departement Landbouw en Visserij, afdeling Monitoring en Studie, Brussel

Diakoulaki, D., Grafakos, S. (2004) ExternE-pol, Externalities of energy: Extension of accounting framework and policy applications, Final report on WP 4. National Technical University Athens, Greece

Dickinson, N.M., Pulford, I.A. (2005) Cadmium phytoextraction using short-rotation coppice *Salix*: the evidence trail. Environ. Int. 31, p. 609-613

Dietz, S. (2007) Review of DEFRA paper: "The Social Cost of Carbon and the Shadow Price of Carbon: what they are, and how to use them in Economic Appraisal in the UK". Review comments

Ding, G.K.C. (2005) Developing a multicriteria approach for the measurement of sustainable performance. Building Res. Inform. 33(1), p. 3-16

Do Nascimento, C.W.A., Xing, B. (2006) Phytoextraction: a review on enhanced metal availability and plant accumulation. Sci. Agric. 63, p. 299-311

Dogliotti, S., Rossing, W.A.H., van Ittersum, M.K. (2004) Systematic design and evaluation of crop rotations enhancing soil conservation, soil fertility and farm income: a case study for vegetable farms in south Uruguay. Agric. Syst. 80, p. 277-302

Dorlhac de Borne, F., Elmayan, T., Roton, C. D., Hys, L. D., Tepfer, M. (1998) Cadmium partitioning in transgenic tobacco plants expressing a mammalian metallothionein gene. Mol. Breed. 4, p. 83-90

Dornburg, V., Faaij, A.P.C. (2005) Cost and CO_2 emission reduction of biomass cascading: methodological aspects and case study of SRF poplar. Clim. Change 71, p. 373-408

Dower, R.C. (1990) Hazardous wastes. In: P.R. Portney (Eds.). Public policies for environmental protection. Resources for the future, Washington DC, p. 151-194

Dries, V. (2007) Werkgroep landbouw. Paper gepresenteerd op het symposium van Benekempen, Januari 2007, Mol

Dushenkov, V., Kumar, P.B.A.N., Motta, H., Raskin, I. (1995) Rhizofiltration: the use of plants to remove heavy metals from aqueous streams. Environ. Sci. Technol. 29, p. 1239-1245

Ebbs, S.D., Lasat, M.M., Brady, D.J., Cornish, J., Gordon, R., Kochian, L.V. (1997) Phytoextraction of cadmium and zinc from a contaminated soil. J. Environ. Qual. 26, p. 1424–1430

Ekins, P. (2007) Peer review by Paul Ekins of DEFRA’s paper. The social cost of carbon and the shadow price of carbon: what they are, and how to use them in economic appraisal in the UK

Envirochem (2005) Identifying environmentally preferable uses for biomass resources Stage 2 report: Life-Cycle GHG Emission Reduction Benefits of Selected Feedstock-to-Product Threads. Vancouver, Canada

EPA (Environmental Protection Agency) (2000a) Introduction to phytoremediation. EPA, Ohio

EPA (2000b) Guidelines for preparing economic analyses. EPA, Ohio

EPA (2008) Green remediation: Incorporating sustainable environmental practices into remediation of contaminated sites. Available from http://cluin.org/greenremediation

EPA (2010) Superfund Green Remediation Strategy. Available from http://www.epa.gov/superfund/greenremediation

Ericsson, K., Rosenqvist, H., Ganko, E., Pisarek, M., Nilsson, L. (2006) An agro-economic analysis of willow cultivation in Poland. Biomass Bioenerg. 30, p. 16-27

Eriksson, O., Finnveden, G., Ekvall, T., Bjorklund, A. (2007) Life cycle assessment of fuels for district heating: a comparison of waste incineration, biomass- and natural gas combustion, Energy Policy 35, p. 1346-1362

Ernst, W.H.O. (1998) The origin and ecology of contaminated, stabilized and non-pristine soils. In: Vangronsveld, J,, Cunningham, S. (Eds.). Metal-contaminated soils: in situ inactivation and phytorestoration. Springer, New York, p. 17–25

Eurelectric/Ecoba (2006) Joint Eurelectric/Ecoba Briefing: Case Study for the Commission's Communication on By-products and for Guidance on the Revised Waste Framework Directive, European Coals Combustion Products Association and Union of the Electricity Industry

European Commission (2001) Ambient air pollution by As, Cd and Ni compounds. Position Paper. Office for official publications of the European Communities, Luxembourg

European Commission (2005a) How to support renewable electricity in Europe? An assessment of the different support schemes. Directorate-General for Energy and Transport, Brussels

European Commission (2005b) Meer doen met minder. Groenboek inzake energie-efficiëntie. Directorate-General for Energy and Transport, Brussels

European Commission (2006a) Soil Protection. The story behind the strategy. EC, Belgium

European Commission (2006b) The new programming period 2007-2013. Guidance on the methodology for carrying out cost-benefit analysis, working document no 4. EC, Brussels

European Commission (2008) Health Check of the CAP, current situation, Commission proposal and Council outcome. Available from ec.europa.eu/agriculture/healthcheck/index_en.htm

European Commission (2009) Part III: Annexes to impact assessment guidelines. EC, Belgium

Eyre, N. (1997) External costs: what do they mean for energy policy? Energy Policy 25(1), p. 85-95

Fankhauser, S., McCoy, D. (2000) Impact analysis of environmental policy. In: H. Folmer, H.L.Gabel (Eds.). Principles of environmental and resource economics. Edward Elgar, Northampton, p. 202-227

Flemish Government (Policy domain Agriculture and Fisheries) (2005) Koolzaad, het nieuwe goud? Teelttechniek van koolzaad. Flemish Ministery, Brussels. Available from
http://www2.vlaanderen.be/landbouw/downloads/plant/koolzaad.pdf.

Federal Public Service Economy: General direction on Statistics and economic information (FOD Economy: statistics) (2006) Landbouwstatistieken: landbouwtelling 2005. Flemish Ministery, Brussels. Available from http://statbel.fgov.be/pub/d5/p501y2005_nl.pdf. Cited 9 July 2009

Federal Public Service Economy: General Direction on Statistics and Economic Information (FOD Economy: statistics) (2007) Landbouwstatistieken: Driemaandelijkse publicatie 2007/1. Flemish Ministery, Brussels

Firbank, L.G. (2008) Assessing the ecological impacts of bioenergy projects. Bioenerg Res. 1, p. 12-19

Fraissler, G., Jöller, M., Mattenberger, H., Brunner, T., Obernberger, I. (2009) Thermodynamic equilibrium calculations concerning the removal of heavy metals from sewage sludge ash by chlorination. Chem. Eng. Process. 48, p. 152–164

French, C.J., Dickinson, N.M., Putwain, P.D. (2006) Woody biomass phytoremediation of contaminated brownfield land. Environ. Pollut. 141, p. 387-395

Fullerton, D., Stavins, R.N. (1998) How economists see the environment. Nature 395, p. 6701

Garbisu, C., Alkorta, I. (2001) Phytoextraction: a cost-effective plant-based technology for the removal of metals from the environment. Bioresour Technol. 77, p. 229-236

GAVE (2005) Op (de) weg met pure plantenolie? De technische, milieu-hygiënische en kostengerelateerde aspecten van plantenolie als voertuigbrandstof. Report 2GAVE-05.05. SenterNovem, Utrecht

Ghekiere, G., Calus, A., Schellekens, A., Van de Ven, G., Van Looveren, M., Chavez, B., De Vliegher, A. (2008) Praktijkonderzoek rassen biogas maïs 2007

Ghosh, M., Singh, S.P. (2005) A review on phytoremediation of heavy metals and utilization of its byproducts. Appl. Ecol. Environ. Res. 3, p. 1-18

Goor, F., Davydchuk, V., Ledent, J-F. (2001) Assessment of the potential of willow SRC plants for energy production in areas contaminated by radionuclide deposits: methodology and perspectives. Biomass Bioenerg. 21, p. 225–235

Goor, F., Davydchuk, V., Vandenhove, H. (2003) GIS-based methodology for Chernobyl contaminated land management through biomass conversion into energy—a case study for Polessie, Ukraine. Biomass Bioenerg. 25, p. 409 – 421

Goossens, S. (2007) Warmte-krachtkoppeling met droge vergisting van energiegewassen: haalbaarheidsstudie. VLAO Directie, Oost-Vlaanderen

Goovaerts, L., Van der Linden, A., Moorkens, I., Vrancken, K. (2009) Beste beschikbare technieken (BBT) voor verbranding van hernieuwbare brandstoffen.

Studie uitgevoerd door het Vlaams Kenniscentrum voor Beste Beschikbare Technieken (VITO) in opdracht van het Vlaams Gewest

Graves, P.E. (2007) Environmental economics, A critique of benefit-cost analysis. Rowman & Littlefield Publishers, Inc., Lanham

Grispen, V.M.J., Nelissen, H.J.M., Verkleij, J.A.C. (2005) Phytoextraction with *Brassica napus L.*: A tool for sustainable management of heavy metal contaminated soils. Environ. Pollut. 144, p. 77-83

Guitouni, A., Martel, J-M. (1998) Tentative guidelines to help choosing an appropriate MCDA method. Eur. J. Oper. Res. 109, p. 501-521

Gustavsson, L. Borjesson, P., Johansson, B., Svenningsson, P. (1995) Reducing CO_2 emissions by substituting biomass for fossil fuels. Energy 20(11), p. 1097-l 113

Hahn, R.W., Stavins, R.N. (1992) Economics of the environment, Economic incentives for environmental protection: integrating theory and practice. AEA papers and proceedings 82(2), p. 464-468

Hall, D.O., Scrase, J.I. (1998) Will biomass be the environmentally friendly fuel of the future? Biomass Bioenerg. 15(4/5), p. 357-367

Hanegraaf, M. C., Biewinga, E. E., van der Bijl, G. (1998) Assessing the ecological and economic sustainability of energy crops. Biomass Bioenerg. 15(4/5), p. 345-355

Hanley, N. (2000) Cost-benefit analysis. In: H. Folmer, H.L. Gabel (Eds.). Principles of environmental and resource economics. Edward Elgar, Northampton, p. 104-129

Hernández-Allica, J., Becerril, J.M., Garbisu, C. (2008) Assessment of the phytoextraction potential of high biomass crop plants. Environ. Pollut. 152, p. 32-40

Hogervorst, J., Plusquin, M., Vangronsveld, J., Nawrot, T., Cuypers, A., Van Hecke, E., Roels, H.A., Carleer, R., Staessen, J.A. (2007) House dust as possible route of environmental exposure to cadmium and lead in the adult general population. Environ. Res. 103, p. 30–37

Holmgren, K., Amiri, S. (2007) Internalising external costs of electricity and heat production in a municipal energy system. Energy Policy 35, p. 5242-5253

Hughes, E. (2000) Biomass cofiring: economics, policy and opportunities. Biomass Bioenerg. 19, p. 457-465

IEABCC (International Energy Agency, Biomass Combustion and Cofiring) (2004) Techno-economic evaluation of selected decentralized CHP applications based on biomass combustion in IEA partner countries. Final report. IEA Task 32 project, Graz

Ignaciuk, A., Vohringer, F., Ruijs, A., van Ierland, A.C. (2006) Competition between biomass and food production in the presence of energy policies: a partial equilibrium analysis. Energy Policy 34, p. 1127-1138

IPCC (2007) Climate Change 2007: synthesis report. An assessment of the Intergovernmental Panel on Climate Change. Cambridge University Press, Cambridge

ITRC (2009) Phytotechnology Technical and Regulatory Guidance and Decision Trees, revised. PHYTO-3. Interstate Technology & Regulatory Council, Phytotechnologies Team, Tech Reg Washington, D.C.

Janikowski, R., Kucharski, R., Sas-Nowosielska, A. (2000) Multi-criteria and multi-perspective analysis of contaminated land management methods. Environ. Monit. Assessm. 60, p. 89-102

Janulis, P. (2004) Reduction of energy consumption in biodiesel fuel life cycle. Renewable Energy 29, p. 861–871

Japenga, J., Koopmans, G.F., Song, J., Römkens, P.F.A.M. (2007) A feasibility test to estimate the duration of phytoextraction of heavy metals from polluted soils. Int. J. Phytorem. 9, p. 115-132.

Jensen, J.K., Holm, P.E., Nejrup, J., Larsen, M.B., Borggaard, O.K. (2009) The potential of willow for remediation of heavy metal polluted calcareous urban soils. Environ. Pollut. 157, p. 931-937

Johansson, P.O. (1987) The economic theory and measurement of environmental benefits. Cambridge University Press, London

Joubert, A.R., Leiman, A., de Klerk, H.M., Katua, S., Aggenbach, J.C. (1997) *Fynbos* (fine bush) vegetation and the supply of water: comparison of multi-criteria decision analysis and cost-benefit analysis. Ecol. Econ. 22, p. 123-140

Kaltschmitt, M., Reinhardt, G.A., Stelzer, T. (1997) Life cycle analysis of biofuels under different environmental aspects. Biomass Bioenerg. 12(2), p. 121-134

Kamm, B., Kamm, M. (2004) Principles of biorefineries. Appl. Microbiol. and Biotechnol. 64(2), p. 137-145

Kayser, A., Wenger, K., Keller, A., Attinger, W., Felix, HR., Gupta, SK., Schulin, R. (2000) Enhancement of phytoextraction of Zn, Cd, and Cu from calcareous soil: the use of NTA and sulfur amendments. Environ. Sci. Technol. 34, p. 1778–1783

Keeney, R.L. (2006) Using preferences for multi-attributed alternatives. J. Multi-Crit. Decis. Anal. 14, p. 169-174

Kelman, S. (1981) Cost-benefit analysis: an ethical critique. AEI J. Gov. Soc. Regul. Jan/Feb, p. 33-40

Keoleian, G.A., Volk, T.A. (2005) Renewable energy from willow biomass crops: life cycle energy, environmental and economic performance. Crit. Rev. Plant Sci. 24, p. 385-406

Khadam, I.M., Kaluarachchi, J.J. (2003) Multi-criteria decision analysis with probabilistic risk assessment for the management of contaminated ground water. Environ. Impact Assess. Review 23, p. 683-721

Kool, A., Timmerman, M., De Boer, H., Van Dooren, H.J., Van Dun, B., Tijmensen, M. (2005) Kennisbundeling co-vergisting. Centrum voor Landbouw en Milieu, Culemborg

Kool, A. (2006) Co-vergisting De Marke. Centrum voor Landbouw en Milieu, Culemborg

Koopmans, G.F., Römkens, P.F.A.M., Fokkema, M.J., Song, J., Luo, Y.M., Japenga, J. and Zhao, F.J. (2008) Feasibility of phytoextraction to remediate cadmium and zinc contaminated soils. Environ. Pollut. 156, p. 905-914.

Kopp, R.L., Abrahamson, L.P., White, E.H., Volk, T.A., Nowak, C.A., Fillhart, R.C. (2001) Willow biomass production during ten successive annual harvests. Biomass Bioenerg. 20, p. 1-7

Kotchen, M. (2010) Cost-benefit analysis, Encyclopedia of climate and weather. Second edition (Ed. S. Schneider). Oxford University press, New York

Krewitt, W. (2002) External costs of energy- do the answers match the questions? Looking back at 10 years of ExternE. Energy Policy 30, p. 839-848

Krutilla, J.V. (1967) Conservation reconsidered. Am. Econ. Rev. 57(4), p. 777-786

Kumar, P.B.A.N., Dushenkov, V., Motto, H., Raskin, L. (1995) Phytoextraction: the use of plants to remove heavy metals from soils. Environ. Sci. Technol. 29, p. 1232–1238

Kuzovkina, Y.A., Quigley, M.F. (2005) Willows beyond wetlands: uses of *Salix L*. species for environmental projects. Water, Air, Soil Pollut. 162, p. 183–204

Lahdelma, R., Salminen, P., Hokkanen, J. (2000) Using multi-criteria methods in environmental planning and management. Environ. Manage. 26(6), p. 595-605

Landberg, T., Greger, M. (1994) Can heavy metal tolerant clones of *Salix* be used as vegetation filters on heavy metal contaminated land? In: Aronsson P, Perttu K (Eds.). Willow vegetation filters for municipal wastewaters and sludges. A biological purification system. Report 50. Swedish Univ Agric Sci, Uppsala, p. 133–144

Langen, M. (2003) Technik der Altholzaufbereitung, presentation on Entsorga congres 4 February 2003, Cologne. Available from http://www.htp.eu/scripte/31_HTP-Langen_Februar-2003.pdf

Ledin, S. (1996) Willow wood properties, production and economy. Biomass Bioenerg. 11(2/3), p. 75-83

Lemmens, B., Ceulemans, J., Elslander, H., Vanassche, S., Brauns, E., Vrancken, K. (2007) Beste Beschikbare Technieken (BBT) voor mestverwerking. Derde editie. VITO, Academia Press, Gent

Lewandowski, I., Schmidt, U., Londo, M., Faaij, A. (2006) The economic value of the phytoremediation function – assessed by the example of cadmium remediation by willow (*salix spp*). Agr. Syst. 89, p. 68-89

Liang, H.M., Lin, T.H., Chiou, J.M., Yeh, K.C. (2009) Model evaluation of the phytoextrion potential of heavy metal hyperaccumulators and non-hyperaccumulators. Environ. Pollut. 157, p. 1945-1952.

Licht, A.L., Isebrands, J.G. (2005) Linking phytoremediated pollutant removal to biomass economic opportunities. Biomass Bioenerg. 28, p. 203-218

LNE (2007) Milieubaten of milieuschadekosten- waarderingsstudies in Vlaanderen. Vlaamse Overheid, Brussel

Loehman, E.T., Randhir, T.O. (1999) Alleviating soil erosion/pollution stock externalities: alternative roles for government. Ecol. Econ. 30, p. 29-46

Lofgren, K.-G. (2000) Markets and externalities. In: H. Folmer, H.L. Gabel (Eds.). Principles of environmental and resource economics. Edward Elgar, Northampton, p. 3-33

Løken, E. (2007) Use of multicriteria decision analysis methods for energy planning problems. Renew. Sust. Energy Rev. 11, p. 1584-1595

Lombi, E., Zhao, F.J., Dunham, S.J. and McGrath, S.P. (2001) Phytoremediation of Heavy Metal–Contaminated Soils: Natural Hyperaccumulation versus Chemically Enhanced Phytoextraction. J. Environ. Qual. 30, p. 1919-1926

Longo, A., Markandya, A., Petrucci, M. (2008) The internalization of externalities in the production of electricity: Willingness to pay for the attributes of a policy for renewable energy. Ecol. Econ. 67, p. 140-152

Machado, P. A., Fu, H., Kratochvil, R. J., Yuan, Y., Hahm, T.-S., Sabliov, C. M., Lo, Y. M. (2010) Recovery of solanesol from tobacco as a value-added byproduct for alternative applications. Bioresour. Technol. 101(3), p. 1091-1096

Madlener, R., Antunes, C.H., Dias, L.C. (2009) Assessing the performance of biogas plants with multi-criteria and data envelopment analysis. Eur. J. Oper. Res. 197(3), p. 387-391

Maeng, H., Lund, H. and Hvelplund, F. (1999) Biogas plants in Denmark: technological and economic developments. Appl. Energy. 64, p. 195-206.

Maksimovic, I., Kastori, R., Krstic, L. and Lukovic, J. (2007) Steady presence of cadmium and nickel affects root anatomy, accumulation and distribution of essential ions in maize seedlings. Biol. Plant. 3, p. 589-592.

Marchaim, U. (1992) Biogas Processes for Sustainable Development, FAO, Rome. Available from www.fao.org/docrep/t0541e/t0541e00.htm

Marchiol, L., Assolari, S., Sacco, P., Zerbi, G. (2004) Phytoextraction of heavy metals by canola (*Brassica Napus*) and radish (*Raphanus Sativus*) grown on multicontaminated soil. Environ. Pollut. 132, p. 21-27

Markandya, A., Harou, P., Bellu, L.G., Cistulli, V. (2002) Environmental Economics for Sustainable growth. A handbook for practicioners. Edward Elgar Publishing, Cheltenham

Mattenberger, H., Fraissler, G., Brunner, T., Herk, P., Hermann,L., Obernberger, I. (2008) Sewage sludge ash to phosphorus fertiliser: Variables influencing heavy metal removal during thermochemical treatment. Waste Manage. 28, p. 2709–2722

Matthews, R.W. (2001) Modelling of energy and carbon budgets of wood fuel coppice systems. Biomass Bioenerg. 21, p. 1–19

Maxted, A.P., Black, C.R., West, H.M., Crout, N.M.J., McGrath, S.P. and Young, S.D. (2007a) Phytoextraction of cadmium and zinc by *Salix* from soil historically amended with sewage sludge. Plant Soil. 290, p. 157-172

Maxted, A.P., Black, C.R., West, H.M., Crout, N.M.J., McGrath, S.P. and Young, S.D. (2007b) Phytoextraction of cadmium and zinc from arable soils amended with sewage sludge using *Thlaspi caerulescens* development of a predictive model. Environ. Pollut. 150, p. 363-372.

McGrath, S., Sidoli, C., Baker, A,, Reeves, R. (1993) The potential for the use of metal-accumulating plants for the in situ decontamination of metal-polluted soils. In: Eijsackers H, Hamers T (Eds.). Integrated soil and sediment research: a basis for proper protection. Kluwer Academic, Dordrecht, The Netherlands, p. 673–676

McGrath, S.P., Zhao, F.J., Lombi, E. (2001) Plant and rhizosphere processes involved in phytoremediation of metal-contaminated soils. Plant Soil 232, p. 207-214

Meers, E., Ruttens, A., Hopgood, M., Lesage, E., Tack, F.M.G. (2005a) Potential of *Brassic rapa*, *Cannabis sativa*, *Helianthus annuus* and *Zea mays* for phytoextraction of heavy metals from calcareous dredged sediment derived soils. Chemosphere 61, p. 561-572

Meers, E., Lamsal, S., Vervaeke, P., Hopgood, M., Lust, N., Tack, F.M.G., Verloo, M.G. (2005b) Availability of heavy metals for uptake by *Salix viminalis* on a moderately contaminated dredged sediment disposal site. Environ. Pollut. 137, p. 354-364

Meers, E., Tack, F.M.G., Meiresonne, L., Ruttens, A., Vangronsveld, J. (2006) Combining biomass production with phytoextraction of Cd and Zn on moderately contaminated sites. Biomass for Energy conference, 25-26 September, Bruges, Belgium

Meers, E., Van Slycken, S., Ruttens, A., Meiresonne, L., Vangronsveld, J., Thewys, T., Tack, F. (2007a) The use of energy crops for phytoremediation allows economic valorisation of moderately contaminated land during site decontamination. ICOBTE Conference. 15-19 July 2007, Beijing, China, p. 915-916

Meers, E., Vandecasteele, B., Ruttens, A., Vangronsveld, J., Tack, F.M.G. (2007b) Potential of five willow species (*Salix spp.*) for phytoextraction of heavy metals. Environ. Exp. Bot. 60, p. 57-68

Meers, E., Tack, F.M.G., Van Slycken, S., Ruttens, A., Vangronsveld, J., Du Laing, G. and Verloo, M.G. (2008a) Chemically-assisted phytoextraction: a review of potential soil amendments for increasing plant uptake of heavy metals. Int. J. Phytorem. 10, p. 390-414.

Meers, E., Soetaert, W., Witters, N., Thewys, T., Vandamme, E. (2008b) Face to Face : Stellingname tegen afschaffing subsidies voor eerste generatie biobrandstoffen. Faces 10(1), p. 52.

Meers, E., Van Slycken, S., Adriaensen, K., Ruttens, A., Vangronsveld, J., Du Laing, G., Witters, N., Thewys, T. and Tack, F.M.G. (2010) The use of bio-energy crops (*Zea mays*) for "phytoattenuation" of heavy metals on moderately contaminated soils: a field experiment. Chemosphere 78, p. 35-41.

Meher, K. K., Panchwagh, A. M., Rangrass, S., Gollakota, K. G. (1995) Biomethanation of tobacco waste. Environ. Pollut. 90(2), p. 199-202

Meiresonne, L. (2006) Kansen, mogelijkheden en toekomst voor de populierenteelt in Vlaanderen. Korte-omloophout voor energieproductie: plaats in het Vlaams bosbeleid. Mei 2006. INBO.R.2006.11. In opdracht van INBO, Geraardsbergen

Meiresonne, L., De Somvile, B., Van Slycken, S., Verdonckt., P., Van Houtte, E., Vandekerckhove, B. (2009) Biomassa van korteomloophout: ook iets voor Vlaanderen? Sylva Belgica 116, p. 36-42.

Mench, M., Tancogne, J., Gomez, A., & Juste, C. (1989) Cadmium bioavailability to *Nicotiana tabacum L.*, *Nicotiana rustica L.*, and *Zea mays* grown in soil amended or not amended with cadmium nitrate. Biol. Fertil. Soils. 8(1), p. 48-53

Mench, M., Lepp, N., Bert, V., Schwitzguébel, J.P., Gawronski, S.W., Schröder, P., Vangronsveld, J. (2010) Successes and limitations of phytotechnologies at field scale: outcomes, assessment and outlook from COST Action 859. J. Soils Sediments 10, p. 1039-1070

Merkhofer, M.W. (1987) Decision science and social risk management, a comparative evaluation of cost-benefit analysis, decision analysis, and other formal decision-aiding approaches. D. Reidel Publishing Company, Dordrecht

Meuleman, B., Faaij, A. (1997) Het gedrag van zware metalen in kolencentrales en afvalverbrandingscentrales, studie uitgevoerd in opdracht van Milieudienst Amsterdam, The Netherlands

Ministerie van de Vlaamse Gemeenschap (2004) Milieubeleidsplan 2003-2007 (MINA 3), Brussel

Ministerie van de Vlaamse Gemeenschap (2008) Actualisatie Milieubeleidsplan 2003-2007 voor de periode 2008-2010 (MINA 3+), Brussel

MIRA (2006) Milieurapport Vlaanderen, Achtergronddocument 2006, Verspreiding van zware metalen. Vlaamse Milieumaatschappij. Available from www.milieurapport.be

MIRA (2008a) Kernset Milieudata MIRA-T 2008, Energiegebruik per energiebron en per (deel)sector in PJ (Vlaanderen, 1990-2007). Available from www.milieurapport.be/nl/feiten-cijfers/

MIRA (2008b) Kernset Milieudata MIRA-T 2008, Emissie van broeikasgassen in kton CO_2-equivalenten (Vlaanderen, 1990-2007). Available from www.milieurapport.be/nl/feiten-cijfers/

Mirasgedis, S., Diakoulaki, D. (1997) Multicriteria analysis vs externalities assessment for the comparative evaluation of electricity generation systems. Eur. J. Oper. Res. 102, p. 364-379

Mirck, J., Isebrands, J.G., Verwijst, T., Ledin, S. (2005) Development of short-rotation willow coppice systems for environmental purposes in Sweden. Biomass Bioenerg. 28, p. 219-228

Mitchell, C.P., Stevens, E.A., Watters, M.P. (1999) Short-rotation forestry – operations, productivity and costs based on experience gained in the UK. Forest Ecol. Manage. 121, p. 123-136

Mulligan, C.N., Yong, R.N., Gibbs, B.F. (2001) Remediation technologies for metal-contaminated soils and groundwater: an evaluation. Eng. Geol. 60, p. 193-207

Munda, G. (1996) Cost-benefit analysis in integrated environmental assessment: some methodological issues. Ecol. Econ. 19, p. 157-168

Murphy, J.D., McKeogh, E. (2006) The benefits of integrated treatment of wastes for the production of energy. Energy. 31, p. 294-310.

Narodoslawsky, M., Obernberger, I. (1996) From waste to raw material-the route from biomass to wood ash for cadmium and other heavy metals. J. Hazard. Mater. 30, p. 157-168

Nawrot, T., Plusquin, M., Hogervorst, J., Roels, H.A., Celis, H., Thijs, L., Vangronsveld, J., Van Hecke, E., Staessens, J.A. (2006) Environmental exposure to cadmium and risk of cancer: a prospective population-based study. Lancet Oncol. 7, p. 119-126

Nawrot, T., Van Hecke, E., Thijs, L., Richart, T., Kuznetsova, T., Jin, Y., Vangronsveld, J., Roels, H.A., Staessen, J.A. (2008) Cadmium-related mortality and long-term secular trends in the cadmium body burden of an environmentally exposed population. Environ Health Persp. 116, p. 1620-1628

Nelson, N., Baker, S., Levine, S.L., Young, L., O'Connor, J., Hill, R.D., Sarofim, A., Wilson, D.G. (1987) Cleanup of contaminated sites. In: L.B. Lave and A.C. Upton (Eds.). Toxic chemicals, health, and the environment. The John Hopkins University Press, Baltimore, p. 205-279

Nielsen, P., Kenis, C., Vanassche, S., Vrancken, K. (2008) Beste Beschikbare Technieken (BBT) voor behandeling van bodemas van huisvuilverbranding. Academia Press, Gent

Nielsen, P. (2008) Actualisatie inzet alternatieven ter vervanging van primaire oppervlaktedelfstoffen Eindrapport. Academie Press, Gent

Nussbaumer, T., Oser, M. (2004) Evaluation of Biomass Combustion based Energy Systems by Cumulative Energy Demand and Energy Yield Coefficient. International Energy Agency IEA Bioenergy Task 32 and Swiss Federal Office of Energy, Verenum press, Version 1.0, Zurich

Obernberger, I., Biedermann, F., Widmann, W., Riedl, R. (1997) Concentrations of inorganic elements in biomass fuels and recovery in the different ash fractions. Biomass Bioenerg. 12(3), p. 211-224

Obernberger, I. (1998) Decentralized biomass combustion: state of the art and future development. Biomass Bioenerg. 13(1), p. 33-56

OECD (2001) Multifunctionality. Towards an analytical framework. Agriculture and food. OECD, France

OVAM Openbare Vlaamse Afvalstoffen Maatschappij (Public Waste Agency of Flanders) (2004a) Basic information for risk assessment – Part 3-H – Formulary of Vlier-Humaan. OVAM, Mechelen

OVAM (2004b) Uitvoeringsplan Houtafval 2004-2008. OVAM, Mechelen

OVAM (2004c) Basisinformatie voor risico-evaluaties – Deel 1-H - Werkwijze voor het opstellen van bodemsaneringsnormen. OVAM, Mechelen

OVAM (2005) BeNeKempen actorenanalyse. Eindrapport. OVAM, Mechelen

OVAM (2006) Voortgangsrapportage 2004-2005. Uitvoeringsplan Organisch Biologisch Afval. OVAM, Mechelen

OVAM (2008a) Vergelijking wet- en regelgeving, beleid en procedure fiches. OVAM, Mechelen

OVAM (2008b) Uitvoeren van milieumetingen voor het bepalen van de externe belasting aan zware metalen in de Noorderkempen – Eindrapport. OVAM, Mechelen

OVAM (2008c) Tarieven en capaciteiten voor storten en verbranden - Actualisatie tot 2007, evolutie en prognose. OVAM, Mechelen

OVAM (2010) Openbare Vlaamse Afvalstoffen Maatschappij (Public Waste Agency of Flanders). Inventarisatie biomassa 2007-2008 (deel 2009) met potentieel 2020. OVAM, Mechelen

Owen, A.D. (2006) Renewable energy: externality costs as market barriers. Energy Policy 34, p. 632-642

Page, C.A., Diamond, M.L., Campbell, M., McKenna, S. (1999) Life-cycle framework for assessment of site remediation options: case study. Environ. Toxicol. Chem. 18(4), p. 801-810

Pahl, O., Firth, A., MacLeod, I., Baird, J. (2008) Anaerobic co-digestion of mechanically biologically treated municipal waste with primary sewage sludge - A feasibility study. Bioresour. Technol. 99, p. 3354-3364

Pearce, D.W., Turner, R.K. (1990) Economics of natural resources and the environment. The John Hopkins University Press, Baltimore

Peene, A. (2009) Anaerobe vergisting van fytoremediatie middels het DRANCO proces (OWS). presentatie MIP-project op 5 mei 2009, Belgium

Peeters, K. (2006) Actieplan Cadmium. Flemish Government, Belgium

Platteau, J., Van Bogaert, T., Van Gijseghem, D. (2009) Landbouwrapport 2008. Departement landbouw en visserij, Brussel

Pollard, S.J.T., Brookes, A., Earl, N., Lowe, J., Kearney, T., Nathanail, C.P. (2004) Integrating decision tools for the sustainable management of land contamination. Sci. Total Environ. 325, p. 15-28

Pollard, S.J.T., Davies, G.J., Coley, F., Lemon, M. (2008) Better environmental decision making-recent progress and future trends. Sci. Total Environ. 400, p. 20-31

Porter, M.E., van der Linde C. (1995) Toward a new conception of the environment-competitiveness relationship. J. Econ. Persp. 9(4), p. 97-118

Price, R., Thornton, S., Nelson, S. (2007) The Social Cost Of Carbon And The Shadow Price Of Carbon: What They Are, And How To Use Them In Economic Appraisal In The UK. DEFRA, London

Provoost, J., Cornelis, C., Swartjes, F. (2006) Comparison of Soil Clean-up Standards for Trace Elements Between Countries: Why do they differ?. J. Soils Sediments 6(3), p. 173 – 181

Pulford, I.D., Watson, C. (2003) Phytoremediation of heavy metal-contaminated land by trees—a review. Environ. Int. 29, p. 529–540

Ramage, J., Scurlock, J. (1996) Biomass, In: Renewable energy, power for a sustainable future (Boyle, G., Ed.). Oxford, Oxford University Press. p. 137-182

Randall, A. (2002) Valuing the outputs of multifunctional agriculture. Eur. Rev. agric. Econ. 29(3), p. 289-307

Raskin, I., Ensley, B.D. (2000) Phytoremediation of toxic metals: using plants to clean up the environment. John Wiley & sons, inc., New York

Raven, R.P.J.M., Gregersen, K.H. (2007) Biogas plants in Denmark: successes and setbacks. Renew. Sust. Energy. Rev. 11, p. 116-132.

Refsgaard, K., Halberg, N., Kristensen, E.S. (1998) Energy Utilization in Crop and Dairy Production in Organic and Conventional Livestock Production Systems. Agric. Syst. 57(4), p. 599-630

Rivers, N., Jaccard, M. (2006) Choice of environmental policy of learning by doing. Energy Econ. 28, p. 223-242

Robinson, B.H., Leblanc, M., Petit, D., Brooks, R.R., Kirkman, J.H., Gregg, P.E.H. (1998) The potential of *Thlaspi caerulescens* for phytoremediation of contaminated soils. Plant Soil 203, p. 47–56

Robinson, B., Fernàndez, J-E., Madejón, T., Murillo, J.M., Green, S., Clothier, B. (2003) Phytoextraction: an assessment of biogeochemical and economic viability. Plant soil 249, p. 117-125

Rosenqvist, H., Roos, A., Ling, E., Hektor, B. (2000) Willow growers in Sweden. Biomass Bioenerg. 18, p. 137-145

Rosenqvist, H., Dawson, M. (2005) Economics of willow growing in Northern Ireland. Biomass Bioenerg. 28, p. 7-14

Ruijgrok, E.C.M., Brouwer, R., Verbruggen, H. (2004) Waardering van natuur, water en bodem in maatschappelijke kosten-batenanalyses. Ministerie van verkeer en waterstaat, The Netherlands

Rulkens, W.H., Tichy, R., Grotenhuis, J.T.C. (1998) Remediation of polluted soil and sediment: perspectives and failures. Water Sci. Technol. 37, p. 27-35

Ruttens, A., De Vocht, A., Vangronsveld, J. (2004) Cadmiummetingen in nieren en vlees van runderen uit de regio Noord-Limburg en de Antwerpse Kempen. Studie uitgevoerd in opdracht van de Openbare Afvalstoffenmaatschappij (OVAM)

Ruttens, A., Vangronsveld, J. (2006) Programme Difpolmine, diffuse pollution from mine activity, realisation. ADEME. Available from http://www.difpolmine.org/servlet/KBaseShow?m-3&cid=10169&catid=10170&sort=-1

Ruttens, A. (2008) Nadere toelichting bij de landbouwgewassen koolzaad en wintertarwe. Presentatie 4de gebruikerscommissie CLO, 20 juni 2008, Balen

Ruttens, A., Vangronsveld, J., Meiresonne, L., Van Slycken, S., Meers, E., Tack, F. (2008) Sustainable use of metal contaminated agricultural soils: cultivation of energy crops as an alternative for classical agriculture. Report by order of OVAM

Ruttens, A., Boulet, J., Weyens, N., Smeets, K., Adriaensen, K., Meers, E., Van Slycken, S., Tack, F., Meiresonne, L., Thewys, T., Witters, N., Carleer, R., Dupae, J., Vangronsveld, J. (2011) Short rotation coppice culture of willow and poplar as energy crops on metal contaminated agricultural soils. Int. J. Phytorem. Accepted for publication

Sadok, W., Angevin, F., Bergez, J-E., Bockstaller, C., Colomb, B., Guichard, L., Reau, R., Dore, T. (2008) Ex ante assessment of the sustainability of alternative cropping systems: implications for using multi-criteria decision-aid methods. A review. Agron. Sustain Dev. 28, p. 163-174

Saez, R.M., Linares, P., Leal, J. (1998) Assessment of the externalities of biomass energy, and a comparison of its full cost with coal. Biomass Bioenerg. 14(5/6), p. 469-478

Salt, D.E., Blaylock, M., Kumar, N.P.B.A., Dushenkov, V., Ensley, B.D., Chet, I., Raskin, I. (1995) Phytoremediation: A novel strategy for the removal of toxic metals from the environment using plants. Biol. 13, p. 468-474.

Salt, D.E., Smith, R.D., Raskin, I. (1998) Phytoremediation. Annu. Rev. Plant Physiol. Plant Mol. Biol. 49, p. 643-668

Sami, M., Annamalai, K., Wooldridge, M. (2001) Co-firing of coal and biomass fuel blends. Prog Energy Combust. Sci. 27, p. 171–214

Samuelson, P.A. (1954) The pure theory of public expenditure. Rev. Econ. Stat. 36(4), p. 387-389.

Sas-Nowosielska, A., Kucharski, R., Małkowski, E., Pogrzeba, M., Kuperberg, J.M., Kryński, K. (2004) Phytoextraction crop disposal—an unsolved problem. Environ. Pollut. 128, p. 373–379

Saxena, P., Krishnaraj, S,, Dan, T., Perras, M., Vettakkorumakankav, N. (1999) Phytoremediation of heavy metals contaminated and polluted soils. In: Prasad M.N.V, Hagemaer J. (Eds.). Heavy metal stress in plants: from molecules to ecosystems. Springer, Berlin, p. 305–329

Scape (2010) Soil conservation and protection in Europe. The way ahead. Available from http://eusoils.jrc.ec.europa.eu/esdb_archive/eusoils_docs/other/SCAPEbook.pdf

Schlamadinger, B., Apps, M., Bohlins, F., Gustavsson, L., Jungmeier, G. Marland, G., Pingoud, K., Savolainen, I. (1997) Towards a standard methodology for greenhouse gas balances of bioenergy systems in comparison with fossil energy systems. Biomass Bioenerg. 13(6), p. 359-375

Schnoor, J.L., Licht, L.A., Mccutcheon, S.C., Wolfe, N.L., Carreira, L.H. (1995) Phytoremediation of Organic and Nutrient Contaminants. Environ. Sci. Technol. 29(7), p. A318-A323

Schwartz, C., Echevarria, G., Morel, J.L. (2003) Phytoextraction of cadmium with *Thlaspi caerulescens*. Plant Soil 249, p. 27–35

SCF (1995) Reports of the Scientific Committee for Food, 36th series, Opinion of the Scientific Committee for Food on cadmium. Available from http://ec.europa.eu/food/fs/sc/scf/reports/scf_reports_36.pdf

Scholz, V., Ellerbrock, R. (2002) The growth productivity and environmental impact of the cultivation of energy crops on sandy soil in Germany. Biomass Bioenerg. 23, p. 81-92

Selvam, A., Wong, J. W-C (2009) Cadmium uptake potential of *Brassica napus* co-cropped with *Brassica parachinensis* and *Zea mays*. J. of Hazard. Mater. 167, p. 170–178

Senternovem (2006) Kansen voor duurzame co-vergisting. Zuidelijke land-en tuinbouworganisatie, Stuurgroep landbouwinnovatie Noord-Brabant, The Netherlands

Shi, G., Cai, Q. (2009) Cadmium tolerance and accumulation in eight potential energy crops. Biotechnol. Adv. 27, p. 555–561

Schmidhuber, J. (2006) Impact of an increased biomass use on agricultural markets, prices and food security: A longer-term perspective. Available from http://www.globalbioenergy.org/uploads/media/0704__Schmidhuber_-_Impact_of_an_increased_biomass_use_on_agricultural_markets__prices_and_food_security.pdf

Sheehan, J, Camobreco, V., Duffield, J., Graboski, M., Shapouri, H. (National Renewable Energy Laboratory) (1998) Life Cycle Inventory of Biodiesel and Petroleum Diesel for Use in an Urban Bus. Final Report Prepared for U.S. Department of Energy's Office of Fuels Development and U.S. Department of Agriculture's Office of Energy

Shi, G., Cai, Q. (2009) Cadmium tolerance and accumulation in eight potential energy crops. Biotechnol. Adv. 27, p. 555–561

Singhal, V., Rai, J.P.N. (2003) Biogas production from water hyacinth and channel grass used for phytoremediation of industrial effluents. Bioresour. Technol. 86, p. 221-225

Skovgaard, M. (2008) CBAs and LCAs: learning from Nordic best practice. Waste Management World. Available from http://waste-management-world.com

Smolders, E., Jansson, G., Van Laer, L. Ruttens, A., Vangronsveld, J., Römkens, P., De Temmerman, L., Waegeneers, N., Bries, J. (2007) Teeltadvies voor de landbouw in kader van het Interreg project BeNeKempen. OVAM, Mechelen

Solino, M., Prada, A., Vazquez, M.X. (2009) Green electricity externalities: forest biomass in an Atlantic European Region. Biomass Bioenerg. 33, p. 407-414

Solow, R.M. (1991) Sustainability: an economist's perspective in: Economics of the environment: selected readings. Second edition. W.W. Norton, New York, p. 179-187

Solow, R.M. (1992) An almost practical step toward sustainability. An invited lecture on the occasion of the 40th anniversary of resources for the future, Washington DC, October 8, 1992

Stavins, R.N. (1997) Economic incentives for environmental regulation. Prepared for The New Palgrave Dictionary of Economics and the Law (ed. P. Newman), 14p.

Stavins, R.N. (1998) What can we learn from the grand policy experiment? Lessons from SO_2 allowance trading. J. econ. Persp. 12(3), p. 69-88

Stavins, R.N., Wagner, A.F., Wagner, G. (2002) Interpreting sustainability in economic terms: dynamic efficiency plus intergenerational equity. Available from www.feem.it/web/activ/_activ.html

Suenens, B. (2007) Verwerking koolzaad op bedrijfsniveau. Eindverhandeling voorgedragen tot het behalen van de graag indusrieel ingenieur in de biotechniek, bachelor in agro- en biotechnologie, landbouw-akkerbouw. Katholieke Hogeschool Kempen, campus Geel

Sulser, T.B., Duryea, M.L., Frolich, L.M., Guevara-Cuaspud, E. (2001) A field practical approach for assessing biophysical sustainability of alternative agricultural systems. Agric. Syst. 68, p. 113-135

Sunstein, C.R. (2005) Cost-benefit analysis and the environment. Ethics 115, p. 351-385

Susarla, S., Medina, V.F., McCutcheon, S.C. (2002) Phytoremediation: an ecological solution to organic chemical contamination. Ecol. Eng. 18, p. 647-658

Suthersan, S.S. (1999) Ch 10: Phytoremediation in: Remediation engineering: design concepts. CRC Press LLC, Boca Raton, 11 p.

Tack, F.M.G., Verloo, M.G. (1996) Metal contents in stinging nettle (*Urtica dioica L.*) as affected by soil characteristics. Sci. Total Environ. 192, p. 31-39.

Tack, F.M.G., Verloo, M.G., Vanmechelen, L., Van Ranst, E. (1997) Baseline concentration levels of trace elements as a function of clay and organic carbon contents in soils in Flanders (Belgium). Sci. Total Environ. 201, p. 113-123.

Tahvanainen, L., Rytkonen, V.-M. (1999) Biomass production of *salix viminalis* in southern Finland and the effect of soil properties and climate conditions on its production and survival. Biomass Bioenerg. 16, p. 103-117

Tahvonen, O., Kuuluvainen, J. (2000) The economics of natural resource utilization. In: H. Folmer, H.L. Gabel (Eds.). Principles of environmental and resource economics. Edward Elgar, Northampton, p. 665-699

Theunis, J., Van der Linden, A., Torfs, R., Vercalsteren, A., Spirinckx, C., Jacobs, A., Vrancken, K. (2003a) Energetische valorisatie van hoogcalorische afvalstromen in Vlaanderen. Deel 2: Afvalaanbod, procesbeschrijvingen en toepassingsmogelijkheden, studie uitgevoerd in het kader van de BBT/EMIS referentieopdracht

Theunis, J., Van der Linden, A., Nielsen, P. (2003b) Energetische valorisatie van hoogcalorische afvalstromen in Vlaanderen: uitstoot van zware metalen. Studie uitgevoerd in opdracht van OVAM, Mechelen

Thewys, T., Witters, N., Ruttens, A., Van Slycken, S., Meers, E., Tack, F.M.G., Vangronsveld, J. (2010a) Economic viability of phytoremediation of an agricultural area using maize: impact on the farmer’s income. Int. J. Phytorem. 12(7), p. 650-662

Thewys, T., Witters, N., Meers, E., Vangronsveld, J. (2010b) Economic viability of phytoremediation of an agricultural area using maize: economics of anaerobic digestion of heavy metal contaminated maize in Belgium. Int. J. Phytorem. 12(7), p. 663-679

Thornley, P., Cooper, D. (2008) The effectiveness of policy instruments in promoting bioenergy. Biomass Bioenerg. 32, p. 903-913

Tietenberg, T. (2003) Environmental and natural resource economics. Sixth edition, Addison Wesley, Boston

Timmerman, M., van Dooren, H.J.C. and Biewenga, G. (2005) Praktijkrapport varkens 42, Mestvergisting op boerderijschaal. Animal Sciences Group, Lelystad, The Netherlands

Tol, R. (2008) The Social Cost of Carbon: Trends, Outliers and Catastrophes. Econ. 2, 2008-25, 24 p.

Torfs, R., De Nocker, L., Schrooten, L., Aernouts, K., Liekens, I. (2005) Internalisering van externe kosten voor de productie en de verdeling van elektriciteit in Vlaanderen. Studie uitgevoerd in opdracht van de Vlaamse Milieumaatschappij, MIRA, Belgium

Torrijos, M., Thalla, A.K., Sousbie, P., Bosque, F., Delgenès, J.P. (2008) Anaerobic digestion of residues from production and refining of vegateble oils as an alternative to conventional solutions. Water Sci. Technol. 5(9), p. 1871-1878

Turner, R.K. (2000) Waste management. In: H. Folmer, H.L. Gabel (Eds.). Principles of environmental and resource economics. Edward Elgar, Northampton, p. 700-744

United Nations (UN) (1998) Kyoto Protocol to the United Nations framework convention on climate change. Available from http://unfccc.int/resource/docs/convkp/kpeng.pdf

UN (2006) The 2nd UN World Water Development Report: Water, a shared Responsibility. United Nations Educational, Scientific and Cultural Organization, Paris. Available from http://www.unesco.org/water/wwap/wwdr/wwdr2/

Van Bogaert, T. (2009) Hoofdstuk 1 Landbouw en voedsel in de wereld. In Platteau, J., Van Bogaert, T., Van Gijseghem, D. (Eds.). Landbouwrapport 2008. AMS, Brussel, p. 26-39

Van Broekhoven, E., Somers, L., Tacquenier, B. (2009) Overzicht van de boekhoudkundige resultaten van 732 land- en tuinbouwbedrijven Boekjaar 2007. Landbouwmonitoringsnetwerk, departement landbouw en visserij, afdeling monitoring en studie

Van-Camp. L., Bujarrabal, B., Gentile, A-R., Jones, R.J.A., Montanarella, L., Olazabal, C., Selvaradjou, S-K. (2004) Reports of the Technical Working Groups Established under the Thematic Strategy for Soil Protection. EUR 21319 EN/4, Office for Official Publications of the European Communities, Luxembourg

Van de Plas, T. (2007) De Belgische biobrandstofindustrie: De impact van het Europees biobrandstofbeleid op de Belgische biobrandstofindustrie, Universiteit Antwerpen

Van den bergh, C. (2009) Biochar and waste law: a comparative analysis. Eur. Energy Environ. Law Rev. October 2009, p. 243-253

Van der Grift, B., and Griffioen, J. (2008) Modelling assessment of regional groundwater contamination due to historic smelter emissions of heavy metals. J. Contam. Hydrol. 96, p. 48-68

Vande Walle, I., Van Camp, N., Van de Casteele, L., Verheyen, K., Lemeur, R. (2007a) Short-rotation forestry of birch, maple, poplar and willow in Flanders (Belgium) I. Biomass production after 4 years of tree growth. Biomass Bioenerg. 31, p. 267-275

Vande Walle, I., Van Camp, N., Van de Casteele, L., Verheyen, K., Lemeur, R. (2007b) Short-rotation forestry of birch, maple, poplar and willow in Flanders (Belgium) II. Energy production and CO_2 emission reduction potential. Biomass Bioenerg. 31, p. 276–283

Van Ginneken, L., Meers, E., Guisson, R., Ruttens, A., Tack, F.M.G., Vangronsveld, J., Diels, L., Dejonghe, W. (2007) Phytoremediation for heavy metal contaminated soils and combined bio-energy production. J. Environ. Eng. Landsc. Manag. 4, p. 227-236

Van Nevel, L., Mertens, J., Oorts, K. and Verheyen, K. (2007) Phytoextraction of metals from soils: how far from practice? Environ. Pollut. 150, p. 34-40

Van Slycken, S., Witters, N., Meers, E., Peene, A., Michels, E., Adriaensen, K., Ruttens, A., Vangronsveld, J., Du Laing, G., Thewys, T. Tack, F.M.G. (xxxx) Safe use of metal contaminated agricultural land by cultivation of energy maize (*Zea mays*). Environ. Pollut., submitted

Van Wezel, A.P., Franken, R.O.G., Drissen, E., Versluijs, C.W., van den Berg, R. (2007) Maatschappelijke Kosten-Baten Analyse van de Nederlandse bodemsaneringsoperatie. MNP rapport 500122002/2007

Vanaken, N. (2008) Mogelijkheden van houtstromen en wettelijke klassering. Workshop ODE, Hout als biomassa voor hernieuwbare energie, 2 December 2008, Belgium

Vandenhove, H., Goor, F., O'Brien, S., Grebenkov, A., Timofeyev, S. (2002) Economic viability of short rotation coppice for energy production for reuse of caesium-contaminated land in Belarus. Biomass Bioenerg. 22, p. 421-431

Vangronsveld, J., Van Assche, F., Clijsters, H. (1995) Reclamation of a bare industrial area, contaminated by non-ferrous metals: in situ metal immobilization and revegetation. Environ. Pollut. 87, p. 51-59

Vangronsveld, J. , Herzig, R., Weyens, N., Boulet, J., Adriaensen, K., Ruttens, A., Thewys, T., Vassilev, A., Meers, E., Nehnevajova, E., van der Lelie, D., Mench, M. (2009) Phytoremediation of contaminated soils and groundwater: lessons from the field, COST Action 859. Environ. Sci. Pollut. Res. 16, p. 765-794.

Vassilev, A., Vangronsveld, J., Yordanov, I. (2002) Cadmium phytoextraction: present state, biological backgrounds and research needs. Bulg. J. Plant Physiol. 28, p. 68–95

Vassilev, A., Schwitzguébel, J.-P., Thewys, T., van der Lelie, D., Vangronsveld, J. (2004) The use of plants for remediation of metal contaminated soils. The Sci.World J. 4, p. 9-34

Vatn, A., Bromley, D.W. (1997) Externalities-A market model failure. Environ. Res. Econ. 9, p. 135-151

Vegter, J.J. (2001) Sustainable contaminated land management: a risk-based land management approach. Land Contam. Reclam. 9(1), p. 95-100

Verma, V.K., Singh, Y.P., Rai, J.P.N. (2007) Biogas production from plant biomass used for phytoremediation of industrial wastes. Bioresour. Technol. 98(8), p. 1664-1669

Verstraete, W. (1981) Biogas. Stichting Leefmilieu v.z.w., Antwerpen, Belgium

Viscusi, W.K., Aldy, J.E. (2003) The Value of a Statistical Life: A Critical Review of Market Estimates Throughout the World. J. Risk Uncertainty 27(1), p. 5–76

Vlaamse Milieu Maatschappij (VMM) (2008) Lozingen in de lucht 1990-2008, Emissies per sector, emissies per thema, internationale rapporteringen, internationale ontwikkelingen. VMM, Belgium

Volk, T.A., Verwijst, T., Tharakan, P.J., Abrahamson, L.P., White, E.H. (2004) Growing fuel: a sustainability assessment of willow biomass crops. Front. Ecol. Environ. 2(8), p. 411-418

VREG (2007) Mededeling van de Vlaamse Reguleringsinstantie voor de Elektriciteits- en Gasmarkt met betrekking tot het al dan niet gebruiken van een compenserende kWh-meter voor de compensatie van de in het distributienet geïnjecteerde elektriciteit. VREG, Belgium

Wang, M., Zou, J., Duan, X., Jiang, W., Liu, D. (2007) Cadmium accumulaton and its effects on metal uptake in maize (*Zea mays, L.*). Bioresour. Technol. 98, p. 82–88.

Wesseler, J. (2007) Opportunities (' costs) matter: A comment on Pimentel and Patzek ''Ethanol production using corn, switchgrass, and wood; biodiesel production using soybean and sunflower''. Energy Policy 35, p. 1414–1416

West, T.O., Marland, G. (2002) A synthesis of carbon sequestration, carbon emissions, and net carbon flux in agriculture: comparing tillage practices in the United States. Agric., Ecosyst. Environ. 91, p. 217–232

Weyens, N., van der Lelie, D., Taghavi, S., Newman, L., Vangronsveld, J. (2009a) Exploiting plant–microbe partnerships for improving biomass production and remediation. Trends Biotechnol. 27, p. 591-598

Weyens, N., van der Lelie, D., Taghavi, S., Vangronsveld, J. (2009b) Phytoremediation: plant-endophyte partnerships take the challenge. Current Opinion Biotechnol. 20, p. 248-254

Weyens, N., van der Lelie, D., Artois, T., Smeets, K., Taghavi, S., Newman, L., Carleer, R., Vangronsveld, J. (2009c) Bioaugmentation with Engineered Endophytic Bacteria Improves Contaminant Fate in Phytoremediation. Environ. Sci. Technol. 43(24), p. 9413-9418.

Weyens, N., Truyens, S., Saenen, E., Boulet, J., Dupae, J., van der Lelie, D., Carleer, R., Vangronsveld, J. (2010) Endophytes and their potential to deal with co-contamination of organic contaminants (toluene) and toxic metals (nickel) during phytoremediation. Int. J. Phytorem., online

Witters, N., Van Slycken, S., Ruttens, A., Adriaensen, K., Meers, E., Meiresonne, L., Tack, F.M.G., Thewys, T., Laes, E., Vangronsveld, J. (2009) Short Rotation Coppice for phytoremediation of a Cd- contaminated agricultural area: A sustainability assessment. BioEnerg. Res. 2(3), p. 144-152

Wong M.H., Cheung, Y.H. (1995) Gas production and digestion efficiency of sewage sludge containing elevated toxic metals, Bioresour. Technol. 54(3), p. 261-268

Zalesny, RS, Wiese,A.H., Bauer,E.O., Riemenschneider, D.E. (2009) Ex situ growth and biomass of Populus bioenergy crops irrigated and fertilized with landfill leachate. Biomass Bioenerg. 33, p. 62–69

Zander, P., Kachele, H. (1999) Modeling multiple objectives of land use for sustainable development. Agric. Syst. 59, p. 311-325

Zhang, Y., Banks, C. (2006) A comparison of the properties of polyurethane immobilised sphagnum moss, seaweed, sunflower waste and maize for the biosorption of Cu, Pb, Zn and Ni in continuous flow packed columns. Water Res. 40, p. 788-798

Zhang, H., Dang, Z., Zheng, L.C., Yi, X.Y. (2009) Remediation of soil co-contaminated with pyrene and cadmium by growing maize (*Zea mays L.*). Int. J. Environ. Sci. Technol. 6, p. 249-258.

Zhao, FJ., Lombi, E., McGrath, SP. (2003) Assessing the potential for zinc and cadmium phytoremediation with the hyperaccumulator *Thlaspi caerulescens*. Plant Soil 249, p. 37–43

Zylicz, T. (2000) Goals, principles and constraints in environmental policies. In: H. Folmer, H.L. Gabel (Eds.). Principles of environmental and resource economics. Edward Elgar, Northampton, p. 130-156

Websites
www.balen.be, last visited April 2010
www.biogas-e.be, last visited February 2010

www.carbonpositive.net, last visited April 2010
www.cen.eu, last visited January 2010
www.codexalimentarius.net, last visited May 2010
www.commonforum.eu, last visited January 2010
www.creg.be, last visited April 2010
www.ecn.nl/phyllis, last visited February 2010
www.e-coal.com, last visited March 2010
www.emis.vito.be, last visited April 2010
www.energietech.info/groengas/theorie/opwerken.html, last visited November 2007
www.eper.ec.europa.eu, last visited 5 October 2009
http://www.erfo.info/CEN-TC-335.25.0.html, last visited February 2010
http://europa.eu/legislation_summaries/food_safety/international_dimension_enlargement, last visited May 2010
http://eusoils.jrc.ec.europa.eu/library/jrc_soil/policy/, last visited May 2010
www.hanze.nl, last visited March 2009
www.ieabcc.nl, last visited March 2010
www.ieabioenergy-task38.org, last visited March 2010
www.ilvo.vlaanderen.be, last visited June 2008
www.ipcc.ch, last visited January 2011
www.iso.org, last visited March 2009
www.nyrstar.com, last visited April 2010
www.ovam.be, last visited January 2010
www.overpelt.be, last visited April 2010
www.reuters.com, last visited April 2010
www.vlaco.be, last visited February 2010
http://www.vmm.be/lucht/luchtkwaliteit/luchtvervuilende-stoffen/welke/zware-metalen.html/resultaten-zware-metalen/meetplaats_00WZ01.html last visited 5 October 2009
www.vreg.be, last visited March 2010
www.wervel.be, last visited March 2009

Personal communications

AMS (Flemish Agriculture and Fishery Department, Monitoring and Study)
Aegten (wholesale trade grains, seeds, and fodder)
Biogas- e vzw (platform for anaerobic digestion): K. Sys, F. Velghe
Cogen Vlaanderen vzw (platform for CHP): A. Stroobandt
ENECO (energy supplier): E. Meers
External firms (third party labor on farm)
Federal Public Service (FOD) Economy: SMEs, independent Professions and Energy
Local farmers (Balen, Lommel, Neerpelt, and Overpelt)
LNE Vlaanderen (Flemish Environment, Nature and Energy Department): Claeys, N.
OVAM (Public Waste Agency Flanders): D. Geysen, N. Vanaken, E. Luyten
OVB (Association for entrepreneurs in soil remediation)
SAPPI Lanaken Mill (producer of paper)
School of Forestry & Environmental Studies, Yale University: M. Kotchen, R. Mendelsohn
VMM (Flemish Environment Agency): A. Rosier

Legislation

Besluit van de Vlaamse regering van 6 februari 1991 houdende vaststelling van het Vlaams reglement betreffende de milieuvergunning (Titel I van het VLAREM) (Belgian State Gazette, B.S. 26.01.1991), laatst gewijzigd bij het Besluit van 21 mei 2010 (B.S. 09.07.2010)

Besluit van de Vlaamse regering van 1 juni 1995 houdende algemene en sectorale bepalingen inzake milieuhygiëne (Titel II van het VLAREM), laatst gewijzigd bij het Besluit van 21 mei 2010 (B.S. 09. 07.2010)

Besluit van de Vlaamse regering van 28 maart 2003 betreffende de subsidiëring van de bebossing van landbouwgronden ter uitvoering van de Verordening (EG) nr. 1257/1999 van de Raad van 17 mei 1999 inzake steun voor plattelandsontwikkeling uit het Europees Oriëntatie- en Garantiefonds voor de Landbouw (EOGFL) en tot wijziging en instelling van een aantal verordeningen (B.S. 19.05.2003), n° 2003-1964, p. 27184-27191

Besluit van de Vlaamse regering van 5 december 2003 tot vaststelling van het Vlaamse reglement inzake afvalvoorkoming en –beheer (VLAREA) (B.S. 30.04.2004), laatst gewijzigd bij het Besluit van 13 februari 2009 (B.S. 01.04.2009), n° 2004-1484, p. 35947-36056

Besluit van de Vlaamse regering van 5 maart 2004 inzake de bevordering van de elektriciteitsopwekking uit hernieuwbare energiebronnen (Energy Decision) (B.S. 23.03.2004), laatst gewijzigd bij besluit van 5 juni 2009 (B.S. 16.09.2009), n° 2004-1041, p. 16296-16301

Besluit van de Vlaamse Regering van 7 juli 2006 ter bevordering van de elektriciteitsopwekking in kwalitatieve warmtekrachtinstallaties (CHP Decision) (B.S. 01.12.2006), n° 2006-4835, p. 66753-66765

Besluit van de Vlaamse Regering van 14 december 2007 houdende vaststelling van het Vlaams reglement betreffende de bodemsanering en de bodembescherming (VLAREBO) (B.S. 22.04.2008), n° 2008-1335, p. 21358-21441

Codex: General Standard for contaminants and toxins in food and feed (CODEX Stan 193-1995), 44 p. Available from http://www.codexalimentarius.net/search/advancedsearch.do

Decreet van 2 juli 1981 betreffende de voorkoming en het beheer van afvalstoffen (Waste Decree), laatst gewijzigd bij het Decreet van van 2 aprll 2004 (B.S. 18.05.2004)

Decreet van 28 juni 1985 betreffende de milieuvergunning (B.S. 17.09.1985), laatst gewijzigd bij Decreet van 27 maart 2009 (B.S. 13.05.2009)

Decreet van de Vlaamse overheid van 13 juni 1990: Bosdecreet (Forest Decree) (B.S. 28. 09.1990), n° 100-3183, p. 18504

Decreet van de Vlaamse overheid van 19 mei 2006 houdende diverse bepalingen inzake leefmilieu en energie (B.S. 20.06.2006), n° 2006-2383, p. 31217-31224

Decreet van 27 oktober 2006 betreffende de bodemsanering en de bodembescherming (B.S., 22 januari 2007) (err., B.S., 20 februari 2007), gewijzigd bij het decreet van 21 december 2007 (B.S., 31 december 2007) en het decreet van 12 december 2008 (Soil Decree) (B.S. 4 februari 2009), n° 2007-318, p. 2579-2602

Decreet van de Vlaamse regering van 22 december 2006 houdende de bescherming van water tegen de verontreiniging door nitraten uit agrarische bronnen (Manure Decree) (B.S. 29.12. 2006), n° 2006-5350, p. 76368-76401

Decreet van 8 mei 2009 houdende algemene bepalingen betreffende het energiebeleid (New Energy Decree) (B.S. 07.07.2009), n° 2009-2380, p. 46145-46191

Koninklijk Besluit van 7 januari 1998 betreffende de handel in meststoffen, bodemverbeterende middelen en teeltsubstraten (K.B. Fertilizer) (B.S. 11.06.1998), laatst gewijzigd bij KB van 28 mei 2003 (B.S. 23.07.2003), n° 98-1451, p. 18947-18957

Koninklijk besluit van 4 maart 2005 betreffende de benamingen en de kenmerken van de biobrandstoffen en andere hernieuwbare brandstoffen voor motorvoertuigen en voor niet voor de weg bestemde mobiele machines (K.B. Biofuels) (B.S. 08.03.2005), n° 2005-632, p. 9620-9621

Koninklijk besluit van 10 maart 2006 betreffende koolzaadolie gebruikt als motorbrandstof (B.S. 20.03.2006), n° 2006-1106, p. 16098-16100

Ministerieel besluit van 12 februari 1999 betreffende de handel en het gebruik van producten die bestemd zijn voor het voederen van dieren (Bijlage I) (Fodder Decision) (B.S. 21.04.1999), laatst gewijzigd bij K.B. van 1 juli 2009 (B.S. 27.07.2009)

Ministerieel besluit van 6 oktober 2006 inzake de vastlegging van referentierendementen voor toepassing van de voorwaarden voor kwalitatieve warmtekrachtinstallaties (CHP reference Decision) (B.S. 01.12.2006), n° 2006-36904, p. 66787-66790

Ministerieel besluit van 1 oktober 2007 betreffende bepalingen en minimumstandaard voor de bedrijfseconomische boekhouding in de landbouw dienstig als basis voor de door de Vlaamse overheid gesteunde adviseringssystemen (B.S. 07.11.2007), n° 2007-36915, p. 56601-56605

Omzendbrief RO/2006/01 van 19 mei 2006 betreffende het afwegingskader en de randvoorwaarden voor de inplanting van installaties voor mestbehandeling en vergisting (B.S. 24.10.2006)

Wet van 4 november 1969: Pachtwet (Law on farming lease) (B.S. 25.11.1969)

Wet van 22 juli 2009 houdende verplichting tot bijmenging van biobrandstof in de tot verbruik uitgeslagen fossiele motorbrandstoffen (Law of 22.07.2009) (B.S. 03.08.2009), n° 2009-2722, p. 51920-51924

Commission staff working document: Document accompanying the Communication from the Commission to the Council, The European Parliament, the European Economic and Social Committee and the Committee of the Regions - Thematic Strategy for Soil Protection – Summary of the Impact Assessment, COM(2006)231 final SEC(2006)1165

Commission staff working document: Document accompanying the Communication from the Commission to the Council, The European Parliament, the European Economic and Social Committee and the Committee of the Regions - Thematic Strategy for Soil Protection - Impact assessment of the thematic strategy on soil protection COM(2006)231 final SEC(2006)1165 SEC(2006)0620

Commission staff working document: Accompanying document to the Communication from the Commission to the Council and the European Parliament Renewable Energy Road Map Renewable energies in the 21st century: building a more sustainable future – Impact Assessment COM(2006)848 final, SEC(2006)1719

Communication from the Commission to the Council, the European Parliament, the Economic and Social Committee of the regions - Towards a Thematic Strategy for Soil Protection COM(2002)179

Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the Regions - Thematic Strategy for Soil Protection COM(2006)231

Communication from the Commission to the Council and the European Parliament - Renewable energy road map - Renewable energies in the 21st century: building a more sustainable future COM(2006)0848

Communication from the Commission to the European Parliament, the Council, the European Economic and Social Committee and the Committee of the Regions - 20 20 by 2020 - Europe's climate change opportunity COM(2008)030

Communication from the Commission to the Council, the European Parliament, the European Economic and Social Committee and the Committee of the regions, Taking sustainable use of resources forward: A Thematic Strategy on the prevention and recycling of waste COM(2005)666 final

Decision of 3 May 2000 (2000/532/EC) replacing Decision 94/3/EC establishing a list of wastes pursuant to Article 1(a) of Council Directive 75/442/EEC on waste and Council Decision 94/904/EC establishing a list of hazardous waste pursuant to Article 1(4) of Council Directive 91/689/EEC on hazardous waste (Waste Catalogue). Official Journal O.J. L 226, 06.09.2000, p. 3-33

Decision of 17 July 2000 on the implementation of a European pollutant emission register (EPER) according to Article 15 of Council Directive 96/61/EC concerning integrated pollution prevention and control (IPPC). O.J. L 192, 28.07.2000, p. 36-43

Directive 89/369/EEC of 8 June 1989 on the prevention of air pollution from new municipal waste incineration plants. O.J. L 163, 14.06.1989, p. 32–36

Directive 89/429/EEC of 21 June 1989 on the reduction of air pollution from existing municipal waste-incineration plants. O.J. L 203, 15.07.1989, p. 50–54

Directive 91/676/EEC of 12 December 1991 concerning the protection of waters against pollution caused by nitrates from agricultural sources. O.J. L375, 31.12.1991, p. 1-8

Directive 91/689/EEC of 12 December 1991 on hazardous waste (Hazardous Waste Directive). O.J. L 377, 31.12.1991, p. 20–27

Directive 94/67/EC of 16 December 1994 on the incineration of hazardous waste, O.J. L 365, 31.12.1994, p. 34–45

Directive 96/61/EC of 24 September 1996 concerning integrated pollution prevention and control (IPPC). O.J. L 257, 10.10.1996, p. 26–40

Directive 96/62/EC of 27 September 1996 on ambient air quality assessment and management (Framework Directive Air.*,* O.J. L 296, 21.11.1996, p. 55–63

Directive 99/30/EC of 22 April 1999 relating to limit values for sulphur dioxide, nitrogen dioxide and oxides of nitrogen, particulate matter and lead in ambient air. O.J. L 163, 29.06.1999, p. 41–60

Directive 99/31/EC of 26 April 1999 on the landfill of waste. O.J. L 182, 16.07.1999 p. 1–19

Directive 2000/76/EG of the European Parliament and the Council of 4 December 2000 on the incineration of waste (WI Directive). O.J. L332, 28.12.2000, p. 91-102

Directive 2001/77/EG of the European Parliament and the Council of 27 September 2001 concerning the promotion of electricity produced from renewable energy sources on the internal electricity market. O.J. L 283 27.10.2001, p. 33-40

Directive 2001/80/EG of the European Parliament and of the Council of 23 October 2001 on the limitation of emissions of certain pollutants into the air from large combustion plants (LCP Directive). O.J. L 309, 27.11.2001, p. 1-22

Directive 2002/32/EC of the European Parliament and of the Council of 7 May 2002 on undesirable substances in animal feed. O.J. L 140, 30.05.2002, p. 10-21

Directive 2003/30/EG of the European Parliament and the Council of 8 May 2003 on the promotion of the use of biofuels or other renewable fuels for transport. O.J. L 123, 17.05.2003, p. 42-46

Directive 2003/96/EC of 27 October 2003 restructuring the Community framework for the taxation of energy products and electricity. O.J. L 283, 31.10.2003, p. 51-70

Directive 2004/8/EC of the European Parliament and of the Council of 11 February 2004 on the promotion of cogeneration based on a useful heat demand in the internal energy market and amending Directive 92/42/EEC. O.J. L 52, 21.02.2004, p. 50-60

Directive 2004/35/EC of the European Parliament and of the Council of 21 April 2004 on environmental liability with regard to the prevention and remedying of environmental damage. O.J. L 143, 30.04.2004, p. 56-75

Proposal for a Directive establishing a framework for the protection of soil and for amending Directive 2004/35/EC of 22 September 2006 (Soil Framework Directive)

Directive 2004/107/EC of the European Parliament and of the Council of 15 December 2004 relating to arsenic, cadmium, mercury, nickel and polycyclic aromatic hydrocarbons in ambient air. O.J. L 23, 26.01.2005, p. 3–16

Directive 2006/12/EC of the European Parliament and of the Council of 5 April 2006 on waste (Waste Directive). O.J. L 114, 27.04.2006, p. 9-21

Directive 2008/50/EC of the European Parliament and of the Council of 21 May 2008 on ambient air quality and cleaner air for Europe. O.J. L 152, 11.06.2008, p. 1-44

Directive 2008/98/EC of the European Parliament and of the Council of the European Union of 19 November 2008 on waste and repealing certain Directives (WF Directive). O.J. L 312, 22.11.2008, p. 3-30

Directive 2009/28/EC of the European Parliament and of the Council of 23 April 2009 on the promotion of the use of energy from renewable sources and amending and subsequently repealing Directives 2001/77/EC and 2003/30/EC (RE Directive). O.J. 140, 05.06.2009, p. 16-62

Press release on 23.04.2009. Commission welcomes adoption of climate and energy package (IP/09/628)

Regulation nº 1782/2003 of 29 September 2003 establishing common rules for direct support schemes under the common agricultural policy and establishing certain support schemes for farmers and amending certain Regulations (CAP Regulation). O.J. L 270, 21.10.2003, p. 1-69

Regulation n° 854/2004 of the European Parliament and of the Council of 29 April 2004 laying down specific rules for the organisation of official controls on products of animal origin intended for human consumption. O.J. L 226, 25.06.2004, p. 83-127

Regulation n° 1881/2006 of 19 December 2006 setting maximum levels for certain contaminants in foodstuffs. O.J. L 364, 20.12.2006, p. 5-24

Regulation (EC) n° 1013/2006 of the European Parliament and of the Council of 14 June 2006 on shipments of waste (Waste Shipment Regulation). O.J. L 190, 12.07.2006, p. 1–98

Regulation 1242/2008 of 8 December 2008 establishing a Community typology for agricultural holdings. O.J. L 335, 13.12.2008, p. 3-24

Regulation (EC) N° 73/2009 of 19 January 2009 establishing common rules for direct support schemes for farmers under the common agricultural policy and establishing certain support schemes for farmers, amending Regulations (EC) No 1290/2005, (EC) No 247/2006, (EC) No 378/2007 and repealing Regulation (EC) No 1782/2003 (Health Check Regulation). O.J. L 30, 31.01.2009, p. 16-99

List of publications

International peer reviewed journals

- **Witters, N.**, Van Slycken, S., Weyens, N., Meers, E., Tack, F., Vangronsveld, J. (xxxx) Does phytoremediation redeem to the expectations of being a sustainable remediation technology: a case study I: Energy production and carbon dioxide abatement. Biomass Bioenerg. submitted
- **Witters, N.**, Van Slycken, S., Weyens, N., Meers, E., Tack, F., Vangronsveld, J. (xxxx) Does phytoremediation redeem to the expectations of being a sustainable remediation technology: a case study II: economic assessment and policy analysis of CO_2 abatement. Biomass Bioenerg. submitted
- **Witters, N.**, Mendelsohn, R.O., Vangronsveld, J. (xxxx) Phytoremediation: Cost-Benefit Analysis comparing current European legislation and economic human toxicological risk assessment. Hum. Ecol. Risk Assess., to be submitted
- **Witters, N.,** Vangronsveld, J., Van Slycken, S., Meers, E., Ruttens, A., Meiresonne, L., Adriaensen, K., Tack, F.M.G., Laes, E. (xxxx) Sustainable contaminated soil management through phytoremediation? The use of Multi Criteria Decision Analysis and Cost Benefit Analysis for crop choice on the heavy metal enriched agricultural Campine area (BE). J. Environ. Manage., to be submitted
- **Witters, N.**, Vanheusden, B., Van Slycken, S., Tack, F.M.G., Meers, E., Meiresonne, L., Carleer, R., Weyens, N., Adriaensen, K., Vangronsveld, J. (xxxx) Marginal effect of heavy metals on the use of biomass from phytoremediation for energy: interpreting the labyrinth of European and Belgian legislation. J. Environ. Manage., to be submitted
- Van Slycken, S., **Witters, N.**, Meers, E., Peene, A., Michels, E., Adriaensen, K., Ruttens, A., Vangronsveld, J., Du Laing, G., Thewys, T. Tack, F.M.G. (xxxx) Safe use of metal contaminated agricultural land by cultivation of energy maize (*Zea mays*). Environ. Pollut., submitted
- Van Slycken, S., Meers, E., **Witters, N.**, Peene, A., Michels, E., Adriaensen, K., Ruttens, A., Vangronsveld, J., Du Laing, G., Thewys,T., Tack, F.M.G. (xxxx) Energy production by anaerobic digestion of energy maize (*Zea mays spp.*) grown on moderately metal-contaminated soils in the Campine region (Belgium). Sci. Total Environ., submitted
- Van Slycken, S., Croes, S., Guisson, R., **Witters, N.**, Meers, E., Adriaensen, K., Michels, E., Ruttens, A., Vangronsveld, J., De Jonghe, W., Thewys, T., Tack, F.M.G. (xxxx) Exploring the phytoextraction potential of Cadmium and Zinc on metal contaminated agricultural soils using *Brassica napus L.* Environ. Pollut., submitted
- Ruttens, A., Boulet, J., Weyens, N., Smeets, K., Adriaensen, K., Meers, E., Van Slycken, S., Tack, F., Meiresonne, L., Thewys, T., **Witters, N.**, Carleer, R., Dupae, J., Vangronsveld, J. (2011) Short rotation coppice culture of willow and poplar as energy crops on metal contaminated agricultural soils. Int. J. Phytorem., accepted
- Thewys, T., **Witters, N**., Ruttens, A., Van Slycken, S., Meers, E., Tack, F.M.G., Vangronsveld, J. (2010) Economic viability of phytoremediation of an agricultural area using maize: impact on the farmer's income. Int. J. Phytorem. 12(7), p. 650-662

- Thewys, T., **Witters, N**., Meers, E., Vangronsveld, J. (2010) Economic viability of phytoremediation of an agricultural area using maize: economics of anaerobic digestion of heavy metal contaminated maize in Belgium. Int. J. Phytorem. 12(7), p. 663-679
- Compernolle, T., **Witters, N.**, Van Passel, S., Thewys, T. (2010) Analyzing a self -managed CHP system for greenhouse cultivation as a profitable way to reduce CO_2-emissions. Energy, special issue, online
- Meers, E., Van Slycken, S., Adriaensen, K., Ruttens, A., Vangronsveld, J., Du Laing, G., **Witters, N.**, Thewys, T., Tack, F.M.G. (2010) The use of bio-energy crops (*Zea mays*) for 'phytoattenuation' of heavy metals on moderately contaminated soils: a field experiment. Chemosphere 78, p. 35-41
- **Witters, N.**, Van Slycken, S., Ruttens, A., Adriaensen, K., Meers, E., Meiresonne, L., Tack, F.M.G., Thewys, T., Laes, E., Vangronsveld, J. (2009) Short Rotation Coppice for phytoremediation of a Cd-contaminated agricultural area: A sustainability assessment. BioEnergy Research 2(3), p. 144-152

Oral presentations

- Organisation conference: Symposium on sustainable soil management, Yale University, School of Forestry & Environmental Studies, sponsored by the Center for Industrial Ecology, New Haven, USA, 18 February 2010

- **Witters, N.**, Mendelsohn; R.O., Van Slycken, S., Thewys, T., Meers, E., Tack, F.M.G., Vangronsveld, J. (2011) Phytoremediation of metal enriched agricultural soils: an economic optimization tool. 11th international conference on the biogeochemistry of trace elements, Congress Center, Florence, Italy, 3-7 July 2011
- **Witters, N.**, Van Slycken, S., Weyens, N., Thewys, T., Meers, E., Tack, F., Vangronsveld, J. (2011) Is phytoremediation a sustainable remediation technology based on its CO_2 abatement potential? International Symposium on Bioremediation and sustainable environmental technologies, Battelle, Reno, Nevada, 27-30 June 2011
- Meers, E., Van Slycken, S., **Witters, N.**, Meiresonne, L., Adriaensen, K., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2010). Biomass on Contaminated Land (BCL) : de mogelijkheden van non-food houtige teelten voor valorisatie van marginale en verontreinigde gronden. Studiedag houtige biomassa in de landbouw, Universiteit Antwerpen, Dept. Bio-ingenieurswetenschappen, Antwerp, Belgium, 16 December 2010
- Meers, E., Van Slycken, S., **Witters, N.**, Meiresonne, L., Adriaensen, K., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2010). Mogelijkheden van korte-omloophout voor fytoremediatie van verontreinigde bodems. Tweedaagse studiedag omtrent het duurzame hergebruik van hout, Faculteit Bio-ingenieurswetenschappen, Universiteit Gent, Ghent, Belgium, 20 December 2010
- **Witters, N.**, Vanheusden, B., Van Slycken, S., Tack, F.M.G., Meers, E., Meiresonne, L., Carleer, R., Weyens, N., Adriaensen, K., Thewys, T. Vangronsveld, J. (2010) Marginal effect of heavy metals on the use of biomass from phytoremediation for energy: interpreting the labyrinth of

European and Belgian legislation. Contaminated site management in Europe (CSME), Ghent, Belgium, 19-21 October 2010

- Van Slycken, S., Meers, E., **Witters, N.**, Adriaensen, K., Thewys, T., Tack, F.M.G., Vangronsveld, J. Sustainable management of metal-contaminated soils in the Campine region: Phytoremediation and bio-energy production. Presentation at Ghent University for the visiting Professor Dr. Prasad, Ghent, Belgium, 1 October 2010
- **Witters, N.**, Mendelsohn, R.O., Van Slycken, S., Tack, F.M.G., Meers, E., Meiresonne, L., Weyens, N., Adriaensen, K., Ruttens, A., Thewys, T., Vangronsveld, J. (2010) External benefit of phytoremediation: The marginal effect of potential CO_2 abatement on the price of biomass. International Phytotechnologies Conference, Parma, Italy, 26-29 September 2010, p. 68
- Van Slycken, S. **Witters, N.**, Meiresonne, L., Meers, E., Adriaensen, K., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2010) Sustainable management of metal contaminated soils in the Campine region using phytoremediation with short rotation coppice. International Phytotechnologies Conference, Parma, Italy, 26-29 September 2010, p. 79
- Van Slycken, S., **Witters, N.**, Croes, S., Meers, E., Adriaensen, K., Guisson, R., Dejonghe, W., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2010) Verkenning van de toepassing van koolzaad op verontreinigde gronden. Studiedag Phytomanagement van metaal-verontreingigde gronden in de Kempen, Fac. Bio-ingenieurswetenschappen, Lab. Ecochemie, Universiteit Gent, Ghent, Belgium, 22 September 2010
- Meers, E., Van Slycken, S., Adriaensen, K., **Witters, N.**, Thewys, T., Tack, F.M.G., Vangronsveld, J. (2010) Toepassing van energiemais als bio-energiegewas op metaalverontreinigde gronden. Studiedag Phytomanagement van metaal-verontreingigde gronden in de Kempen, Fac. Bio-ingenieurswetenschappen, Lab. Ecochemie, Universiteit Gent, Ghent, Belgium, 22 September 2010
- Van Slycken, S., **Witters, N.**, Meiresonne, L., Meers, E., Adriaensen, K., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2010) Sustainable management of metal contaminated soils in the Campine region using phytoremediation with short rotation coppice. Sustainable approaches to remediation of contaminated land in Europe (SARCLE), Ghent, Belgium, 8-18 June 2010, p. 48
- **Witters, N**., Van Slycken, S. (2010) Economic assessment of phytoremediation: biomass for energy on heavy metal enriched land, Symposium on sustainable land management, Yale University, School of Forestry & Environmental Studies, New Haven, USA, 18 February 2010
- Van Slycken, S., **Witters, N.**, Meers, E., Adriaensen, K., Meiresonne, L., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2009) The use of energy maize (Zea mays) for phytoattenuation of metal-enriched soils in the Campine Region. Contaminated site management in Europe, Ghent, Belgium, 27-29 October 2009, p. 133
- Compernolle, T., **Witters, N.**, Van Passel, S., Thewys, T. (2009) Analyzing a self -managed CHP system for greenhouse cultivation as a profitable way to reduce CO_2-emissions. 5th Dubrovnik conference on sustainable development of energy, water and the environment systems, Dubrovnik, Croatia, 29 September- 3 October 2009

- Vangronsveld, J., Meers, E., Van Slycken, S. Meiresonne, L., Adriaensen, K., **Witters, N.**, Thewys, T., Boulet, J., Weyens, N., van der Lelie, D., Dejonghe, W., Carleer, R., Ruttens, A. (2009) Metal accumulation in plants with added economic value grown on metal contaminated soils: sustainable use of these soils for bio-energy production and possibilities for phytoextraction. 10th international conference on the biogeochemistry of trace elements (ICOBTE), Chihuahua, Mexico, 13-18 July 2009
- **Witters, N.,** Vangronsveld, J., Van Slycken, S., Meers, E., Ruttens, A., Meiresonne, L., Adriaensen, K., Thewys, T., Tack, F.M.G., Laes, E. (2009) Assessment of phytoremediation of a heavy metal contaminated agricultural area: the use of MCDA in addition to CBA, Postgraduate course Energy and Climate, Institute of environment and sustainable development, Antwerp, Belgium, 18 May 2009
- Van Slycken, S., Van Nevel, L., **Witters, N.**, Meiresonne, L., Mertens, J., Vangronsveld, J., Meers, E., Verheyen, K., Tack, F.M.G. (2009) Fytoremediatie van metaalverontreinigde bodems in de Kempen: mogelijkheden en valkuilen, Starters in het bosonderzoek, ANB, INBO, Katholieke Universiteit Leuven, Universiteit Antwerpen, Universiteit Gent, Vrije Universiteit Brussel en Inverde, Brussel, 19 March 2009
- Van Slycken, S., **Witters, N**., Meers, E., Ruttens, A., Meiresonne, L., Kuppens, T., Tack, F.M.G., Vangronsveld, J., Thewys, T. (2009) Energiegewassen op landbouwgronden aangerijkt met zware metalen: onderzoek naar functionaliteitsherstel door middel van fytoextractie en/of biomassateelt als volwaardig alternatief voor klassieke landbouw. Opendeurdag, Faculteit bio-ingenieurswetenschappen, Ghent, Belgium, 14 March 2009
- Van Slycken, S., **Witters, N**., Meers, E., Adriaensen, K., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2009) Energy maize for phytoremediation of metal-enriched soils and production of energy; Case: the Campine region (BE). AgSAP conference, Integrated Assessment of Agriculture and Sustainable Development, Setting the agenda for Science and Policy, Egmond Aan Zee, The Netherlands, 10-12 March 2009, p. 396-399
- **Witters, N.**, Van Slycken, S., Meers, E., Adriaensen, K., Meiresonne, L., Tack, F.M.G., Thewys, T., Vangronsveld, J. (2009) Phytoremediation of a heavy metal contaminated agricultural area combined with energy production. Multifunctional use of energy maize, rapeseed and Short Rotation Crops in the Campine (BE), MACE conference. Green Week Scientific Conference, Multi-level processes of integration and disintegration, Berlin, Germany, 14-15 January 2009, Proceeding of the Third Green Week Scientific Conference (Eds. F. Schaft, A. Balmann), p 200-208
- Van Slycken, S., **Witters, N.**, Meers, E., Meiresonne, L., Adriaensen, K., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2008) How can plants help us to remediate soils and create green energy? Young Botanists' Day, Flora and vegetation of Belgium and neighbouring regions, 2nd circular, Brussels, Belgium, 21 November 2008
- Van Slycken, S., Meers, E., Meiresonne, L., **Witters, N.**, Adriaensen, K., Peene, A., Dejonghe, W., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2008) The use of bio-energy crops for phytoremediation of metal

enriched soils in the Campine region, 14^{th} PhD symposium on applied biological sciences, Ghent, Belgium, 15 September 2008, Communications in Agricultural and Applied Biological Sciences 73(1), p. 19-20

- **Witters, N**., Ruttens, A., Thewys, T. (2008) Multiple criteria deciding on phytoremediation of a heavy metal contaminated agricultural area. Case: The Campine, Belgium. Proceedings of the Short Rotation Crops International Conference: Biofuels, Bioenergy and Bioproducts, Minneapolis (MN), USA, 18-22 August 2008, p. 69
- Meers, E., **Witters, N.**, Ghekière, G., Christiaens, P., Van Slycken, S., Thewys, T., Tack, F.M.G. (2008) Duurzaamheid van biovergisting: een praktijkgetuigenis, Biogas als opstap naar 2e generatie biobrandstoffen, Ghent, Ghent Bioenergy Valley (GBEV)/Biogas-E, 24 June 2008
- Dejonghe, W., Meers, E., Ruttens, A., Guisson, R., Van Ginneken, L., Van Slycken, S., **Witters, N.**, Peene, A., Tack, F.M.G., Vangronsveld, J., Thewys, T., Kegels, J., Beeckman, E., Smis, J., De Schepper, S., Fastenaekels, H., Diels, L. (2008) Phytoremediation for heavy metal contaminated soils and combined bio-energy production. Proceedings of Innovations for Sustainable Production Conference (i-SUP) Bruges, Belgium, 22-25 April 2008
- **Witters, N.**, Thewys, T. (2007) Hernieuwbare energie uit biomassa via vergisting: economisch rendabel? Milieu Innovatie Platform (MIP) event, Hasselt University, Diepenbeek, Belgium, 25 October 2007
- Diels, L., Dejonghe, W., Geurds, M., Guisson, R., Van Ginneken, L., Meers, E., Ruttens, A., **Witters, N.**, Thewys, T., Vangronsveld, J., Kegels, J., Defoort, B., Beeckman, E., Smis, J., De Schepper, S., Fastenaekels, H. (2007) Energy crop production combined with phytoremediation of heavy metal contaminated soils. COST Action 859 – Workshop of WG2 and WG4 and Management Committee Meeting, Vilnius, Lithuania, Fate of pollutants in the plant/rhizosphere system: Fundamental aspects and their significance for field applications - Prospects and research needs, p. 63-64
- Dejonghe, W., Geurds, M., Guisson, R., Van Ginneken, L., Diels, L., Meers, E., **Witters, N.**, Thewys, T., Vangronsveld, J., Kegels, J., Befoort, B., Beeckman, E., Smis, J., De Schepper, S., Fastenaekels, H. (2007) Phytoremediation for heavy metal contaminated soils and combined bio-energy production. 9th ICOBTE Conference, Biogeochemistry of trace elements: environmental protection, remediation and human health, Beijing, China, 15-19 July 2007, p. 162-163

Poster Presentations

- **Witters, N.**, Mendelsohn, R.O., Van Slycken, S., Weyens, N., Thewys, T., Meers, E., Tack, F., Vangronsveld, J. (2011) Phytoremediation: Cost-Benefit Analysis comparing current European legislation and economic human toxicological risk assessment. International Symposium on Bioremediation and sustainable environmental technologies, Battelle, Reno, Nevada, 27-30 June 2011
- **Witters, N.**, Mendelsohn, R.O., Thewys, T., Vangronsveld, J. (2011) Phytoremediation: Cost-Benefit Analysis comparing current European legislation and economic human toxicological risk assessment. 21^{st}

annual conference on soils, sediments, water, and energy, San Diego, USA, 11 March 2011

- **Witters**, N., Van Slycken, S., Adriaensen, K., Meiresonne, L., Meers, E., Tack, F.M.G., Vangronsveld, J., Thewys, T. (2009) Economic viability of phytoremediaation of an agricultural area using energy maize (Zea mays), Contaminated site management in Europe, Ghent, Belgium, 27-29 October 2009, p. 205
- **Witters,** N., Van Slycken, S., Adriaensen, K., Meiresonne, L., Meers, E., Tack, F.M.G., Vangronsveld, J., Thewys, T., Laes, E. (2009) Remediation assessment of a heavy metal contaminated agricultural area: The use of a Multi Criteria Decision Tool in addition to CBA, Contaminated site management in Europe, Ghent, Belgium, 27-29 October 2009, p. 204
- Van Slycken, S., **Witters, N.**, Meiresonne, L., Meers, E., Adriaensen, K., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2009) The Use of Short Rotation Coppice for Phytoremediation and Sustainable Management of Metal-Enriched Soils in the Campine Region, Contaminated site management in Europe, Ghent, Belgium, 27-29 October 2009, p. 202
- Van Slycken, S., Meiresonne, L., **Witters, N**., Meers, E., Adriaensen, K., Vangronsveld, J., Thewys, T., Tack, F.M.G. (2009) Korteomloophout op bodems uit de Kempen aangerijkt met zware metalen. Van Wilg tot Warmte, Demonstratienamiddag Exploitatie Korteomloophout, Rijkevelde, Belgium, 11 February 2009
- Van Slycken, S., **Witters, N.**, Meers, E., Ruttens, A., Meiresonne, L., Kuppens, T., Tack, F.M.G., Vangronsveld, J., Thewys, T. (2008) Energiegewassen op landbouwgronden aangerijkt met zware metalen: onderzoek naar functionaliteitsherstel door middel van fytoextractie en/of biomassateelt als volwaardig alternatief voor klassieke landbouw, BeNeKempen Symposium, OVAM, Turnhout, 23 June 2008
- Van Slycken, S., **Witters, N**., Meers, E., Meiresonne, L., Adriaensen, K., Thewys, T., Vangronsveld, J., Tack, F.M.G. (2008) How can plants help us to remediate soils and create green energy? POWERLINK, Oostende, Belgium, 27 November 2008
- Van Slycken, S., Meers, E., Ruttens, A., Meiresonne, L., **Witters, N.**, Tack, F.M.G., Vangronsveld, J., Thewys, T. (2007) Combining phytoremediation with bioenergy production in the Campine region, Umicore Networking Event, Ghent, Belgium, 20 September 2007
- Van Slycken, S., Meers, E., Ruttens, A., Meiresonne, L., **Witters, N.**, Tack, F.M.G., Vangronsveld, J., Thewys, T. (2007) Korte Omloop Houtteelt op bodems uit de Kempen aangerijkt met zware metalen, Starters in het bosonderzoek, ANB, INBO, Katholieke Universiteit Leuven, Universiteit Antwerpen, Universiteit Gent, Vrije Universiteit Brussel en Inverde, Brussel, 22 March 2007
- Van Slycken, S., Meers, E., Ruttens, A., Meiresonne, L., **Witters, N.**, Tack, F.M.G., Vangronsveld, J., Thewys, T. (2007) Alternatieve teelten op bodems verontreinigd met zware metalen, BeNeKempen Symposium, OVAM, Mol, 25 January 2007

Local Journals

- Schildermans, T., Vanheusden, B., **Witters, N.** (2010) Op zoek naar indicatoren voor een duurzame ontwikkeling van brownfields. Burger, bestuur en beleid 6(1), p. 17-38

- Declercq, S., **Witters, N.**, Thewys, T. (2009) Hoe kan België voldoen aan de doelstelling om 10% biobrandstof tegen 2020 bij te mengen, gegeven Europese richtlijnen en gegeven haar eigen wetgeving? Een kritische sectoriële analyse. Milieu & Economie 2009(4), p. 13-15
- **Witters, N**., Meers, E., Thewys, T. (2008) Vergisting van energiemaïs: economisch en energetisch rendabel. Milieu & Economie 2008(4), p. 18-19
- Compernolle, T., **Witters, N.** (2008) WKK in de glastuinbouw: economisch haalbaar in eigen beheer?. Proeftuinnieuws 18(19), p. 37-38
- Meers, E., **Witters, N**., Soetaert, W., Vandamme, E., Thewys, T. (2008) Biobrandstoffen: feiten en mythes rond het voedsel vs bio-energie debat. De Drietand 2008(28), p. 8
- Meers, E., Soetaert, W., **Witters, N.**, Thewys, T., Vandamme, E. (2008) Face to Face: stellingname tegen afschaffing subsidies voor eerste generatie biobrandstoffen. Faces 10(1), p. 52
- Meers, E., Soetaert, W., Vandamme, E., Verstraete, W., **Witters, N**., Thewys, T. (2008) Milieurendement van biovergisting. Landbouw en Techniek 13(september), p. 40-41
- Janssens, B., Meers, E., Van Slycken, S., Ruttens, A., Vangronsveld, J., Thewys, T., **Witters, N.**, Tack, F.M.G. (2008) Teelt van energiegewassen als alternatief voor de klassieke landbouw. Zinkorama 38, p. 26-27