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Rheinischen Friedrich-Wilhelms-Universität zu Bonn

The relation between indicators for the crediting of emission rights and abatement costs – a systematic modeling approach for dairy farms

I n a u g u r a l – D i s s e r t a t i o n

zur

Erlangung des Grades

Doktor der Agrarwissenschaften
(Dr. agr.)

der
Landwirtschaftlichen Fakultät
der
Rheinischen Friedrich-Wilhelms-Universität
zu Bonn

vorgelegt am 26. Juni 2013
von
Bernd Lengers
aus
Laer

Referent:	Prof. Dr. Karin Holm-Müller
Korreferent:	Prof. Dr. Thomas Heckelei
Tag der mündlichen Prüfung:	08. November 2013
Erscheinungsjahr:	2013

Meinen Eltern

Vorwort

Mein Dank gilt…

Frau Prof. Dr. Karin Holm-Müller für die Förderung und Unterstützung während meiner Promotionszeit, das mir entgegengebrachte Vertrauen, nötige Maß an Freiraum und die übertragene Verantwortung in allen Bereichen meiner Tätigkeit am Institut. Desweiteren gilt mein ganz besonderer Dank Dr. Wolfgang Britz für die enge Zusammenarbeit, Hilfe und Unterstützung während der Modellentwicklung, Klärung methodischer Fragen und Anfertigung gemeinsamer Publikationen. Beiden bin ich sehr dankbar für fruchtbare Diskussionen und die sachliche und ergebnisorientierte Zusammenarbeit im Projekt und darüber hinaus.

Prof. Dr. Thomas Heckelei möchte ich für die Verfügbarkeit und Hilfestellung bei ökonometrischen und publikationstechnischen Fragen als auch für die Übernahme des Korreferats danken.

Auch bei meinen Doktorandenkolleginnen und –kollegen möchte ich mich für die angenehme Arbeitsatmosphäre, weiterführende fachliche Gespräche und offenes Gehör für das ein oder andere Problem bedanken. Dabei vor allem bei meiner Bürokollegin Johanna Garbert für die fachlichen Diskussionen und kritische Durchsicht meiner Publikationsprojekte. Frau Dr. Andrea Zimmermann möchte ich für ihre Hilfestellung bei der Methodenwahl und Klärung einiger ökonometrischer Fragen bei meinem letzten Publikationsprojekt danken.

Frau Inga Schiefler vom Institut für Landtechnik an der Universität Bonn danke ich für die motivierte und motivierende Zusammenarbeit bei zwei gemeinsamen Publikationsprojekten, den kritischen und bereichernden fachlichen Austausch sowie für zahlreiche aufmunternde Tassen Kaffee.

Desweiteren möchte ich meinen Studienfreunden und meinen Freunden aus der Heimat für ihren Zuspruch und ihre Freundschaft danken, auch wenn ich mich aufgrund von räumlichen Disparitäten oft rar gemacht habe.

Nicht zuletzt möchte ich mich für die jahrelange Unterstützung bei meiner Familie bedanken. Bei meinen Geschwistern Hubertus, Martina und Frank und ganz besonders bei meinen Eltern dafür, dass sie mir den Spaß und das Interesse an der Landwirtschaft vermittelt, meine Begeisterung gefördert und erhalten haben und mir in allen Belangen Rückhalt bieten.

Abschließend möchte ich mich von ganzem Herzen bei meiner Freundin Anna-Sophia für die Unterstützung während meiner Promotionszeit, aber vor allem für ihren aufmunternden und belebenden Charakter und die dadurch erwirkte Ablenkung von der Arbeit bedanken.

Kurzfassung

Im Rahmen des Kyoto-Protokolls haben sich 37 Industrienationen dazu verpflichtet ihre Treibhausgasausstöße im Vergleich zum Basisjahr 1990 drastisch zu senken. Um dieses Ziel zu erreichen, wurde in Europa das Emissionshandelssystem (ETS) eingeführt. Bislang sind nur industrielle Sektoren und der Flugverkehr direkte Teilnehmer am ETS. Landwirtschaftliche Treibhausgase (THG) gehen lediglich in das jeweilige nationale Emissionsinventar ein und fallen nicht unter das ETS. In Deutschland lag der Anteil landwirtschaftlicher Treibhausgasemissionen an den Gesamtemissionen im Jahr 2007 bei ca. 5.6%. Nun stellt sich die Frage ob und inwieweit auch landwirtschaftliche Produktionsprozesse kosteneffizient zu Vermeidungsleistungen beitragen können (hier zunächst anhand der Milchviehhaltung). Bei marktbasierten Instrumenten (wie dem ETS) müssen die Emissionsmengen der Teilnehmer jedoch klar quantifizierbar sein. Landwirtschaftliche THGs stammen vornehmlich aus diffusen und somit nicht unmittelbar messbaren Quellen. Es werden daher Berechnungsschemata (THG-Indikatoren) benötigt, die anhand von Betriebsparametern und definierten Emissionsfaktoren das THG-Inventar des Betriebes herleiten. Die Konstruktion dieser Indikatoren bedingt dabei auf welche Prozessinformationen des Betriebes die Berechnungen zurückgreifen und bestimmt dadurch, welche Vermeidungsmöglichkeiten vom Indikator angerechnet werden. Daher ist anzunehmen, dass die Konstruktion des Indikators erheblichen Einfluss auf die *Genauigkeit* der errechneten THG-Inventare, die *Umsetzbarkeit* des Indikators aus einzelbetrieblicher und politischer Sicht, als auch auf die *Realisierung kostengünstiger Vermeidungsstrategien* im landwirtschaftlichen Betrieb hat.

Zur Untersuchung dieser Punkte wurde ein hoch aufgelöstes, dynamisches, gemischt-ganzzahliges lineares Optimierungsmodell konstruiert, welches auf spezialisierte Milchviehbetriebe angepasst ist. Dieses ermöglicht die Ableitung und den Vergleich von betrieblichen Grenzvermeidungskosten (GVK) von THG in Abhängigkeit des genutzten THG-Indikators. Desweiteren wurden Meta-Modelle geschätzt, um die Wirkung von einzelbetrieblichen Charakteristika, Preisen und Gestaltungsmöglichkeiten einer Reduktions-Politik (Reduktionsziel, Vermeidungshorizont, Indikator…) auf die Höhe der GVK zu quantifizieren.

Die Ergebnisse der Arbeit zeigen, dass die Höhe der GVK stark von der Komplexität des gewählten THG-Indikators beeinflusst wird. Vorteile im Bezug auf die induzierten Vermeidungskosten sind bei detaillierten Indikatoren gegeben, da diese die Nutzung von variablen und kostengünstigen Vermeidungsstrategien erlauben. Dies gilt vor allem bei geringen Reduktionsmengen, wobei die Kostenvorteile detaillierter Indikatoren mit steigender Vermeidungsleistung abnehmen. Je detaillierter und komplexer ein Indikator konstruiert ist, desto genauer ist auch die errechnete Emissionsmenge. Die Umsetzbarkeit aus einzelbetrieblicher als auch politischer Sicht nimmt jedoch mit steigender Komplexität der Berechnung ab, da die Erhebung und Kontrolle der Betriebsinformationen aufwändiger wird. Es zeigt sich also ein starker Trade-off zwischen der Umsetzbarkeit und den Aspekten Genauigkeit der Messung und Realisierung kosteneffizienter Vermeidung.

Außerdem gibt es, bedingt durch die Ungenauigkeit der Indikatoren, eine starke Verzerrung zwischen den errechneten und den tatsächlichen GVK. Eine Vielzahl betrieblicher und nicht betriebliche Faktoren haben signifikanten Einfluss auf die entstehenden Vermeidungskosten, was auf eine starke Heterogenität der GVK in der Gesamtpopulation schließen lässt. Das erschwert die Abschätzbarkeit der Vermeidungsleistung und der Kostenbelastung bei einer marktbasierten Einbeziehung der Milchviehhaltung. Desweiteren sind die GVK im Bereich der Milchviehhaltung im Vergleich zu anderen Sektoren relativ hoch und in Anbetracht aktueller CO_2-Preise im ETS nur äußerst geringe Vermeidungsmengen unter Nutzung hoch detaillierten Indikatoren zu erreichen, welche mit hohem administrativem Aufwand verbunden wären. Die Aktivierung möglicher kostengünstiger Vermeidungsleistungen in der Milchviehhaltung ist somit zurzeit nicht effizient durch marktbasierte Instrumente umsetzbar. Gesetzliche Auflagen in Synergie mit schon bestehenden baulichen und umweltpolitischen Auflagen scheinen erfolgversprechender.

Dennoch ist eine tiefere und weiterführendere Analyse der Sachverhalte notwendig. Grundlegend dafür hat die Arbeit gezeigt, dass das umweltpolitische Instrument, die sektoralen Reduktionsziele und vor allem der angewandte THG-Indikator bei Diskussionen um mögliche Vermeidungsmöglichkeiten in der Landwirtschaft zwingend gemeinsam diskutiert werden müssen. Für weitere Analysen bieten die aufgezeigten Ergebnisse als auch die entwickelten Modelle und Analysemethoden gute Ausgangspunkte.

Schlagwörter: Treibhausgase, Emissionsberechnung, Betriebsmodellierung, Milchviehhaltung, THG-Indikatoren, THG-Vermeidungskosten.

Abstract

With the Kyoto Protocol, industrial nations agreed to reduce greenhouse gas (GHG) emissions compared to the 1990 level. Therefore, in Europe, the emissions trading scheme (ETS) was implemented, to date incorporating industrial sectors and aviation. Agriculture is up to now only included in the reporting mechanisms for national GHG inventories. In Germany, the agricultural sector emissions accounted for 5.6% of the national total in 2007. The questions therefore arise, whether and how agriculture may contribute to the national reduction goals. For market-based instruments (like the ETS), the emission inventories of participants need to be quantifiable. However, since most agricultural GHGs originate from diffuse sources, they cannot be measured directly and therefore have to be derived by calculation schemes (indicators) which use farm-level data to approximate the actual emissions. Though, the construction of the indicators determines which farm-level information are accessed for calculation. This directly impacts which GHG mitigation options are accounted by the indicator. Hence, it can be assumed that the indicator construction impacts on the *accuracy* of GHG accounting, the indicators' *feasibility* on single farms and in a political context, as well as the ability to induce *low-cost abatement* strategies on farm level.

To investigate these aspects, a highly detailed single farm, mixed integer linear programming model was developed which is adjustable to a wide range of dairy farm characteristics and a set of promising GHG indicators. It enables single farm abatement and marginal abatement costs (MACs) for the mitigation of GHGs to be derived and compared using different detailed indicators. Further on, a meta-modeling approach was used to derive statistical dependencies between farm characteristics, prices, factors of a potential environmental policy design (targeted reduction level, accounting period, indicator…) and the simulated MACs.

By influencing the ability of choosing abatement measures at a farm level, the indicators show a strong impact on the level of the resulting MACs. Detailed indicators allow for low-cost abatement for low reduction levels, but the cost advantages of detailed indicators level off with increasing abatement amounts. The results indicate that the more detailed the indicator definition is, the higher the GHG accounting accuracy also is. The feasibility of such indicators at a farm level as well as on the political level on the other hand decreases in complexity of indicator definition. A great trade-off was found between feasibility of indicators and the other requirements of measurement accuracy and low-cost abatement. Furthermore, a variety of farm characteristics and prices show significant impacts on the MACs which points to highly heterogeneous MAC structures in the actual farm population. Thus, the predictability of potential abatement amounts and resulting cost burdens in the farm population deteriorates. In addition, MACs in dairy farming are relatively high compared to other sectors and cost effective abatement amounts in a price relevant range (with regard to actual carbon prices) are up to now only achievable with a rather small potential under detailed indicators which induce high administrative burdens. Hence, to date, cost efficient GHG mitigation in dairy farming is hardly achievable under

market-based instruments. Statutory requirements in line with existing building or environmental regulations may be the more meaningful solution and gain synergy effects.

Nevertheless, further, more detailed analyses regarding these issues are required. Whereat, this work has shown that the environmental policy instrument, the sectoral reduction goals, and especially the applied GHG indicator are topics not to be discussed separately in the discussion about possible and cost efficient abatement possibilities in agriculture. Therefore, the obtained results and developed modeling and analytical approaches can deliver good starting points.

Keywords: greenhouse gases, emission accounting, farm modeling, dairy farming, GHG indicators, GHG abatement costs.

Abbreviations

a	annum
ACs	abatement costs
ASMGHG	Agricultural Sector and Greenhouse Gas Mitigation Model
avACs	average abatement costs
BMELV	Bundesministerium für Ernährung, Landwirtschaft und Verbraucherschutz
C	carbon
CAPRI	Common Agricultural Policy Regionalised Impact Modelling System
CH_4	methane
cm	centimeter
CO_2	carbon dioxide
d	day
DAIRYDYN	dairy dynamic
DFG	Deutsche Forschungsgemeinschaft
DM	dry matter
DMI	dry matter intake
DOE	design of experiments
eq.	equation
equ.	equivalents
EC	European Communities
ECM	energy corrected milk
EFEM	Economic Farm Emission model
EPA	Environmental Protection Agency
ERG	Eastern Research Group
ETS	emission trading scheme
EU	European Union
EUC	European Commission
FAO	Food and Agricultural Organization
FDZ	Forschungsdatenzentren der Statistischen Ämter des Bundes der Länder
GAMS	general algebraic modelling system
GE	gross energy
GHG	greenhouse gas

GUI	graphical user interface
GWP	global warming potential
h	hour
ha	hectare
HFC	hydrofluorocarbon
IPCC	Intergovernmental Panel on Climate Change
kg	kilogram
KTBL	Kuratorium für Technik und Bauwesen in der Landwirtschaft
KW	kilowatt
l	liter
LHS	latin hypercube sampling
LKV	Landeskontrollverband
LP	linear programming
LU	livestock unit
LWK	Landwirtschaftskammer
m	meter
MACs	marginal abatement costs
MAPD	mean absolute percentage deviation
MDSM	Moorepark Dairy Systems Model
MIP	mixed integer programming
MPI	Ministry of Primary Industies
N	nitrogen
N_2O	nitrous oxide
OECD	Organisation for Economic Co-operation and Development
OLS	ordinary least squares
PFC	perfluorocarbon
PTFE	polytetrafluoroethylene
RAINS	Regional Air Pollution Information and Simulation model
SF	sulfurhexafluorid
SON	state of nature
SSM	supply side model
t	ton
UBA	Umweltbundesamt
UNFCCC	United Nations Framework Convention on Climate Change
VIF	variance inflation factor
XP	crude protein

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Chapter 1: **Introduction**[1]

1.1 Problem statement

With the Kyoto Protocol, industrial nations agreed to reduce greenhouse gas (GHG) emissions stemming from different industrial and nonindustrial sectors up to 2012 by about 5% relative to the 1990 levels (UNFCCC, 2009; UNFCCC, 2013a). The EU defined ambitious goals for a reduction of 20 to 30%[2] until 2020 relative to the 1990 GHG inventories (UNFCCC, 2013b: p.7). Although currently solely emissions from industrial sectors and the flight sector are regulated by cap-and-trade mechanisms under the Kyoto Protocol, GHGs stemming from agricultural production are included in the reporting mechanism and are hence also incorporated into the overall inventories. New reduction goals for the member states were planned to be enacted at the 2012 UN climate conference in Qatar. The member states are yet to come up with a solution concerning further reduction goals, but have agreed to extend the Kyoto Protocol until 2020 (Kyoto II) and to negotiate a new agreement in 2015.

According to the IPCC (2007), agriculture accounted for 13.5% of global GHG emissions of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) in 2004 stemming from ruminant fermentation, land cultivation, fertilizer practice and further farm processes. Therefore, to reach ambitious GHG emission reduction goals, mitigation potentials in agriculture will also have to be investigated. To date, only New Zealand officially envisages the inclusion of agriculture in its emission trading system (ETS), though indirectly, by allocating GHGs from agriculture to milk and meat processors, live animal exporters and fertilizer companies which are already part of the ETS (MPI, 2013). There are also discussions in the EU and the US to address agricultural emissions with

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[1] This dissertation consists of results that were obtained during the research work on a DFG funded project (HO 3780/2-1) with the title "The relation between indicators for the crediting of emission rights and abatement costs – a systematic modeling approach for dairy farms", supervised by Prof. Dr. Karin Holm-Müller and Dr. Wolfgang Britz from the Institute of Food and Resource Economics, University of Bonn.

[2] "As part of a global and comprehensive agreement for the period beyond 2012, the European Union reiterates its conditional offer to move to a 30 per cent reduction by 2020 compared to 1990 levels, provided that other developed countries commit themselves to comparable emission reductions and developing countries contribute adequately according to their responsibilities and respective capabilities" (UNFCCC, 2013b).

economic instruments like taxes, emission trading or statutory requirements (e.g. ANVEC, 2011; BMELV, 2010; BREEN, 2008; DECARA and VERMONT, 2011; PÉREZ et al., 2009; SCHNEIDER et al., 2008).

Any market based instrument like tradable emission permits or emission taxes leads to a theoretically cost efficient abatement of GHGs, as any emitter will reduce GHG emissions as long as the marginal costs for an additional GHG unit abated (MACs) are below the emission price. Hence, MACs may serve as an important tool for "[...] (1) estimating the economic effects on individual agents of command-and-control abatement instruments and (2) estimating the total mitigation effect of a market-based abatement instrument" (PÉREZ, 2006: p.170). For the application of such emission regulation policies, emission inventories of single participants have to be quantified exactly so that the required emission permits or the tax burden are able to be determined. However, the problem with agricultural emissions is that they generally stem from diffuse non-point sources (e.g. soils or farm premises). This renders direct measurements very expensive or even impossible in comparison to other industrial sectors where measurements can be made by the "bottle-neck principle" (e.g. emissions from exhaust chimneys). Thus, monitoring cannot draw on actual emissions, but must be based on accounting rules (GHG indicators) which estimate emissions from observable farm attributes (herd sizes, milk yields, feed use, etc.).

One approach to minimize monitoring costs would hence be an indicator-based assessment using emission coefficients for different agricultural activities (PÉREZ and HOLM-MÜLLER, 2007). Such an approach does not necessarily minimize abatement costs, as shown by HOLM-MÜLLER and ZIMMERMANN (2002) for the general case of a product-oriented emission tax. For agricultural emissions specifically, DECARA and JAYET (2001) point out that uniform emission coefficients per production unit (e.g. per cow) will lead to an inefficient use of abatement measures because of, inter alia, regional differences in emission coefficients.

Policy design thus needs not only to compare the performance of different instruments such as a standard, tax or tradable permits, but must simultaneously decide upon a suitable emission indicator. The choice of a specific indicator might have wide ranging impacts on the efficiency and targeted precision of policy instruments. Agents will only realize abatement options accounted for under a specific indicator, such that cost effective options could be missed and further biases in the aspired policy impacts may occur as the applied GHG indicators always involve a certain degree of emission accounting inaccuracies. The choice of the indicator scheme in environmental policy design hence may have manifold impacts on the resulting abatement and marginal abatement costs for emissions in agriculture which demands a systematic analysis of these aspects.

As existing research disregards the impact of the indicator scheme used for emission quantification and restriction on the farm or the agricultural sector level, this needs a deeper investigation of possible effects on individual farms as well as on societal level. Up to now, only DURANDEAU et al. (2010) have mentioned possible differences in farm-level abatement costs depending on the level of detail offered by the emission quantification scheme (i.e. if GHG calculations of the indicator fall back on highly aggregated or very detailed farm process information). This dissertation aims to close this gap, however, initially choosing the German agricultural dairy sector as an example, as dairy production is the most important agricultural GHG emitter on a global, as well as national scale (4% of global totals stem from dairy farming processes globally (FAO, 2010). LEIP et al. (2010) quantify the portion of dairy production systems for Western Europe to 30% of overall agricultural GHG emissions[3]. In Germany, ¾ of overall livestock GHGs stem from dairy production alone (DÄMMGEN, 2009; UBA, 2009)[4]). Therefore, for any discussion about potential emission reduction efforts in the agricultural context, dairy production will play a decisive role.

1.2 Objective

The main aim of this work is to gain new insights into the influence of the construction of emission indicators on the induced abatement strategies and resulting abatement costs for the farms under regulation. This will give new indications concerning the influence of the construction of GHG indicators on the potential to induce *low-cost abatement* at the farm level. As each GHG indicator should serve as a proxy for actual emissions, the aspect of measurement and abatement *accuracy* will also be investigated in detail. With respect to the fact that the indicators' GHG accounting relies on different detailed farm-level data (which is more or less available in sufficient quality), aspects of the indicators' *feasibility* in an administrative context will also be addressed. Concerning the influence of GHG indicators on the ability to trigger low-cost abatement on farms, this work will also gain

[3] Derived by a cradle-to-gate life cycle assessment under recognition of land use and land use change (LEIP et al., 2010).

[4] Only direct livestock release, no soil and fertilizer emissions. For reference, see chapter 2.

deeper fine grained insights into the dependency of farm attributes, prices or aspects of an environmental policy design on the MAC curves under the use of differently detailed GHG indicator schemes for GHG restriction. This additional knowledge is of relevance for further work and analysis of the MAC distribution (that may vary between indicators) in the actual farm population as it informs on which firm attributes (e.g. factors like herd size, intensity level, manure storage techniques, labor productivity) should be used for systematic upscaling to sectoral or regional MAC curves. The results obtained enable the systematic evaluation of indicators concerning feasibility, accuracy and low-cost abatement, which is necessary for any discussion about the applicability of indicator schemes from a farm level as well as from a political perspective. Hence, results will ultimately allow recommendations to be made on potentials, applicability as well as single farm and sectoral effects of any environmental policy design for the inclusion of dairy farming (or agriculture in general) into GHG reduction plans.

To reach those goals, three related *methodological working objectives* have been formulated:

- Construction of a set of differently detailed GHG calculation schemes which quantify emission inventories based on farm-level information.
- Construction of a highly detailed bio-economic single farm simulation model to simulate single dairy farm reactions to emission ceilings and the resulting GHG mitigation costs under different GHG indicators (representative for real world dairy farms).
- Development of a systematic meta-modeling procedure to statistically analyze outputs of the single farm simulation model.

1.3 Proceeding

The dissertation proceeds as follows[5]:

Chapter 2 provides background information on the overall topic of GHG emissions from agriculture and dairy production. Detailed information are given on the amounts of methane, nitrous oxide and carbon dioxide stemming from different processes on dairy

[5] Chapter 2 to 4 comprise of three technical papers and documentations which were prepared during the work on the DFG funded project (HO 3780/2-1). Chapter 5 to 8 present four single studies that are already published or submitted to international peer-reviewed journals (referred to in the footnotes of the single chapters). Tables and graphs are numbered separately for each chapter, starting with number 1 at the beginning of each chapter.

farms. The background information about the main sources and the physical and chemical processes the different gases stem from are especially important for the understanding of mitigation potentials of different abatement strategies on the farm level.

To give an overview on the available as well as relevant German GHG abatement options in dairy production, chapter 3 explains single GHG mitigation measures. This identifies the farm-level processes relevant for single options, also discussing flexibility aspects of the measures' applicability over time. This detailed explanation is deemed to be important as these measures impact the ability of each farm to define effective GHG mitigation strategies.

A technical documentation of the construction of the different GHG indicators under investigation in this work is given in chapter 4. The formal illustration of the accounting formulas clarifies the level of accounting detail and relevant farm-level information required for GHG calculation under the different indicators. Additionally, from information on the process variables implemented in the calculations, inferences on the accounted, previously described mitigation measures under the single indicators can be made. This provides the background information for subsequent, more analytical chapters. This chapter also defines a reference indicator (as most appropriate proxy for actual emissions) which is later used to evaluate the GHG accounting accuracy of the other indicators.

Afterwards, chapter 5 describes the model approach of DAIRYDYN (Dairy Dynamic), the highly detailed single dairy farm simulation model on which later analyses are based (supply side model). A deeper insight into the model construction is given with special attention to the methodology of MAC derivation for single simulated farms under different indicators. Preliminary results illustrate that the overall approach leads to proper MAC estimates and gives first indications on differences between MACs under different indicators and especially underlines the importance of a dynamic framework to investigate flexibility aspects of abatement strategies.

To assess the accuracy of the previously developed simulation model and adherent GHG indicator schemes, in chapter 6, estimates from one-year online measurements (only methane compound was available in sufficient quality) of a real life experimental dairy barn on Haus Riswick, Germany are compared with indicator-specific GHG estimates derived by simulations of an identical parameterized farm experiment with the model DAIRYDYN.

Building on previously presented preliminary results and conclusions concerning the indicators feasibility, accuracy and ability to trigger low-cost GHG abatements on a farm as well as on a societal level, chapter 7 comprises a detailed analysis of these three indicator requirements. It is based on simulation results obtained using the model DAIRYDYN. Thus, an evaluation of the different indicators is possible, which also includes a discussion of GHG indicator applicability with respect to the targeted reduction levels. The results enable conclusions to be drawn with regard to a potential inclusion of dairy farming into GHG reduction efforts and relating consequences for an appropriate policy design and the general applicability of the investigated indicators for any indicator-based GHG regulation scheme.

Based on the formerly discovered differences in MACs between different parameterized farms, chapter 8 gives deeper insights into the dependency between farm-level attributes, prices, indicators and other aspects of an environmental GHG regulation scheme on the resulting farm-level MACs. To perform appropriate statistical analyses on the main MAC driving factors, an efficient sampling procedure is constructed to create a representative sample of single farm experiments which are then executed by the model DAIRYDYN. Simulated sample results by DAIRYDYN are then used to derive statistical meta-models which express significant MAC influencing factors on the farm level and their quantitative effect. This allows more microeconomic production oriented results at the farm level to be obtained compared to the existing literature. It hence provides helpful information which can be used to analyze for heterogeneity aspects of MACs if the analysis is to be widened to a regional or sectoral level.

Chapter 9 gives concluding remarks from the results obtained and further hints towards future research activities for which the developed modeling approaches may be appropriate, and fields where further scientific investigations are necessary.

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Chapter 2: **GHG survey of German agriculture – specific view on dairy production systems**[6]

Abstract

This chapter comprises a summary and categorization of different greenhouse gases occurring in different sectors in Germany and from agricultural and dairy production systems in particular. Further specific characterization of different greenhouse gases, including nitrous oxide (N_2O), methane (CH_4) and carbon dioxide (CO_2) is given regarding their primary sources and especially the major processes stemming from in the farming sector which are responsible for and influence their occurrence. In 2007 Germany emitted nearly 1 billion t CO_2-equivalents, of which 5.6% stemmed from agricultural processes.

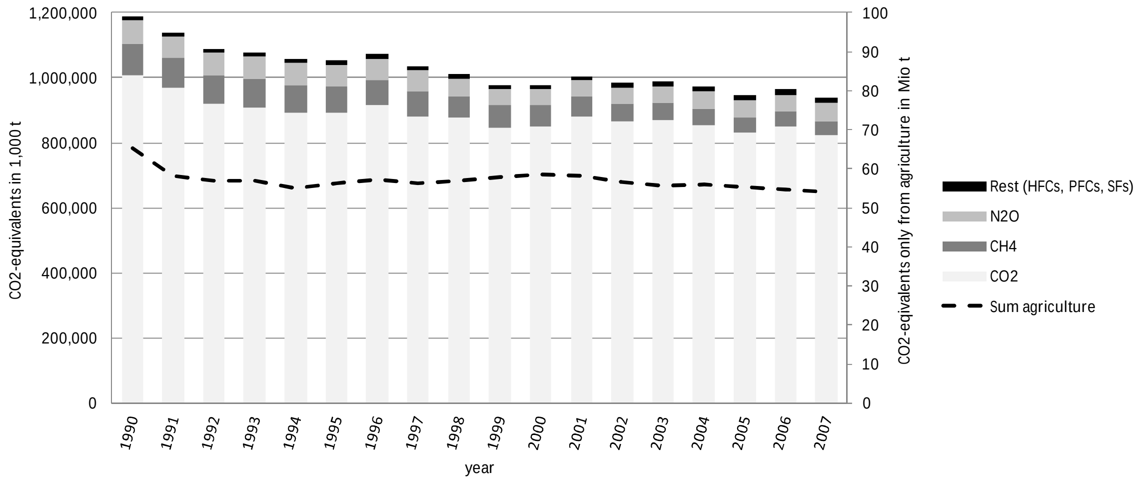
Keywords: *GHG emissions, agriculture, dairy farms.*

[6] This part is based on a technical paper which was developed during the work on the DFG funded project with reference number HO 3780/2-1. The technical paper is available on the project related web-page of the Institute of Food and Resource Economics of the University of Bonn as LENGERS, B. (2011): GHG survey of German agriculture - specific view on dairy production systems. http://www.ilr.uni-bonn.de/agpo/rsrch/dfg-ghgabat/dfgabat_e.htm

2.1 Greenhouse gases in Germany

As a consequence of national efforts to reduce GHG emissions from different sectors in Germany, the overall emission levels of anthropogenic gases declined markedly over recent decades. According to the results of the UMWELTBUNDESAMT[7] (UBA, 2009) and DÄMMGEN (2009), Germany's GHG emissions in 2007 added up to 942,047,000 t CO_2-equ. in total. The following graphic presents the development of overall GHG emissions in CO_2-equ. from all sectors of the German economy. As illustrated in the figure below, total emissions diminished by 20.8% from 1990 to 2007. CO_2 accounts for about 86% of total emissions in each year on average. This GHG is followed by CH_4, with an average yearly fraction of 6.6% and N_2O with about 5.7% of total emissions (expressed in CO_2-equ). The emissions of the remaining GHGs (like HFCs, PFCs, SFs)[8] are negligible, amounting to 1.4% of total emissions on average. Because of these exiguous amounts, stemming mainly from industry emissions, HFCs, PFCs and SFs are not considered further here.

Figure 1: **Total GHG emissions in Germany from 1990–2007 in 1,000t CO_2-equivalents**



Source: own illustration following UBA (2009).

Specific values for the year 2007 are highlighted in table 1, showing amounts of the single basic gases emitted in CO_2-equivalents. Nearly 9/10 of total emissions were direct CO_2 emissions, followed by 5.9% from N_2O and 4.5% CH_4 (expressed in CO_2-equ.).

[7] As this chapter is based on a technical paper from 2011 and inventory calculations of the UBA are retroactively adjusted if UBA calculation algorithms change, the derived inventories and percentage shares may differ with regard to inventory reports of other publication years.

[8] These are the three greenhouse gases (major industrial) - hydrofluorocarbons (HFCs), perfluorocarbons (PFCs), and sulfurhexafluoride (SFs).

Consequently, concerning the relative fraction of each GHG type, no evident change took place comparing their relative values in 2007 with their average fractions over time. But changes in the emission levels of single gases occurred from 1990 until 2007. CO_2 decreased by 18.2%, CH_4 decreased by 56.5% and N_2O declined by 20.8% (UBA, 2009). Weighting these reductions by the average fractions of each gas, it sums up to an overall decline in GHGs of 20.8%, as already stated at the beginning of this chapter.

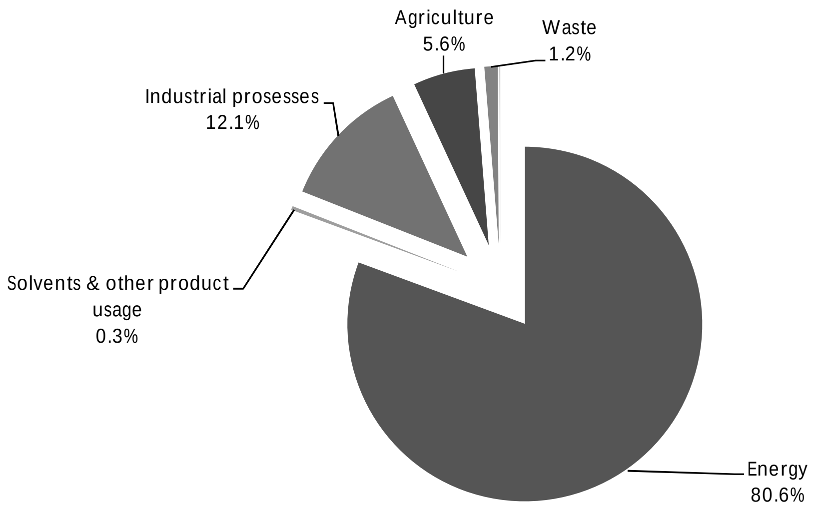
Table 1: **Specific values of GHG emissions of Germany in 2007** (basic gas calculation of Rest not possible because of different global warming potentials of the gases)

Gas	percentage of whole emissions [%]	emission amount of basic gas [t]	emission amount [t CO_2-equ]
CO_2	87.7%	826,424,000	826,424,000
CH_4	4.5%	2,026,286	42,552,000
N_2O	5.9%	180,252	55,878,000
Rest (HFCs, PFCs, SFs)	1.8%	-	17,193,000
Sum			942,047,000

Source: own calculation and illustration following DÄMMGEN (2009) and UBA (2009).

Overall, GHG emissions from Germany added up to 942 Mio. t CO_2-equ. in 2007 as denoted by the table above. These can be allocated to single sectors of the economy congruent to their emission proportions (figure 2).

Figure 2: **Contributions of emitting groups to total German emissions in 2007**



Source: own calculation and illustration following UBA (2009).

As illustrated above, the majority of Germany's GHG emissions in 2007 was determined by energy consumption (80.6%). 12% was produced by industrial processes and the agricultural sector emitted 5.6% of the total GHG in Germany, followed by waste management (1.2%) and solvents and use of other products (0.3%). So, agriculture produced about 54 Mio. t CO_2-equ, with 40.3% stemming from CH_4, 5% from CO_2 and 54.6% caused by direct and indirect nitrous oxide emissions (table 2).

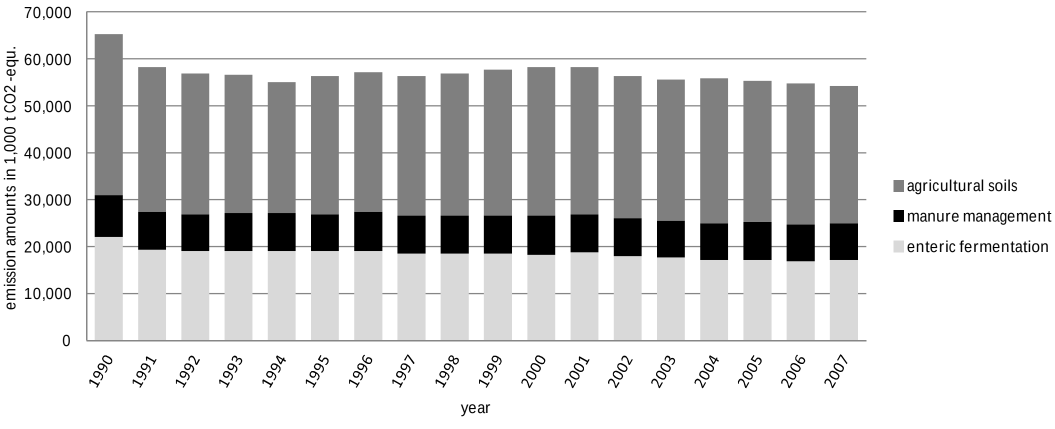
Table 2: **GHGs of German agriculture in 2007 fragmented by gas type and compared with overall German GHGs**

Gas	emission amount [t]	fraction of overall emissions of gas type from all industry sectors[%]	fraction of overall agricultural emissions [%]	emission amount [t CO_2-equ]
CO_2	2,703,000	0.3%	5.0%	2,703,000
CH_4	1,040,118	51.3%	40.3%	21,842,484
N_2O direct	77,292	42.9%	44.2%	23,960,461
N_2O indirect	18,200	10.1%	10.4%	5,642,000
		Sum:	100.0%	54,147,945

Source: own calculation following DÄMMGEN (2009) and UBA (2009).

Overall GHG emissions from agriculture in Germany declined by 17% from 1990 until 2007 as shown by the illustration below. According to table 2, agricultural production is only responsible for 0.3% of German CO_2 emissions. But considering agriculture's fraction of methane and nitrous oxide emissions, agriculture turns out to be a meaningful emitter, accountable for 53% of nitrous oxide (direct + indirect) and 51.3% of German methane emissions in 2007. Even though the proportions of total CH_4 and N_2O in the overall German emissions are relatively low (CH_4: 6.6%; N_2O: 5.7%) it is important to take a closer look at the involved sources in the agricultural sector, because by targeting more than the half of total N_2O as well as CH_4 in Germany can lead to further abatement potentials.

Figure 3: **Production of GHG emissions in 1,000 t CO_2-equ. from German agriculture by source**



Source: own calculations and illustration following DÄMMGEN (2009) and UBA (2009).

With nearly constant emissions from manure management (organic as well as mineral), the 17% decrease in total emissions from agriculture from 1990 to 2007 (see figure 3) can mainly be traced back to reductions in GHGs caused by ruminant fermentation processes (partly due to smaller animal numbers, e.g. from dairy as a main CH_4 emitter, a sector which decreased by 35% in terms of animal numbers during that time period (DÄMMGEN, 2009: p.263)) and varying emissions from agricultural soils (depending on cultivation and climatic impacts).

Apparently, the largest share of each year's agricultural emissions (in CO_2-equ.) stems from agricultural soils (about 54% on average from 1990 to 2007) which were responsible for the highest percentages of whole agricultural N_2O and CO_2 emissions (shown in the fifth column of table 3). Concerning methane emissions, enteric fermentation can be identified as the main culprit. More than ¾ of agricultural methane emissions are caused by ruminant fermentation processes, followed by anaerobic processes in manure during different storage techniques. As seen in the table below, deposition of CH_4 in the soil is also possible, quantifiable to 30,200 t CH_4 in 2007, which, however, reduces overall agricultural emissions only marginally.

Table 3: **GHGs from German agriculture separated according to production source**

source/sink	gas	emission amount [t/a]	emission amount in CO2-equ. [t CO2-equ]	percentage of gas type caused by source
enteric fermentation	CH4	809,285	16,994,985	76%
manure management	CH4	261,033	5,481,699	24%
	N2O	7,752	2,403,061	8%
agricultural soils	CH4	-30,200	-634,200	-3%
	N2O	87,740	27,199,400	92%
	CO2	2,703,000	2,703,000	100%
		Sum:	54,147,945	

Source: own calculations and illustration following DÄMMGEN (2009).

Since agriculture is responsible for more than half the overall German methane and nitrous oxide emissions (table 2), it is important to take a look at the specific sources of these high levels in agriculture. Enteric fermentation processes emitted 809,285 t of CH_4 in 2007, of which 91.7% stemmed from ruminant fermentation of cattle in general. 46.5% can be ascribed to dairy cows alone. Adding fractions of heifers and calves, dairy farm systems (cows, heifers and calves) are responsible for nearly 70% of German methane emissions from digestive activities. Other animal production systems emit only minor amounts of CH_4 due to smaller numbers of animals (like sheep) or the animals being monogastrics. (DÄMMGEN, 2009)

Nearly 50% of methane emissions from manure management were caused by cattle husbandry, directly followed by pigs with 47.3%. Other animal categories were only responsible for percentages of about 1%. Thereby, with regard to manure management as a source, dairy farm systems represented a high fraction of total methane emission, with the amounts from dairy cows, heifers and calves adding up to 40.1% of total manure caused methane.

In the case of nitrous oxide production from manure management activities, the polluter weighting is nearly the same. Cattle farms in total emitted about ¾ of N_2O emissions from animal excreta. Again here, dairy farm systems are primarily liable (64.9%), followed by pigs (19%) and only minor fractions of poultry and other animals.

In the following table, total agricultural emissions of methane, nitrous oxide and carbon dioxide are allocated to the different animal production categories and soils, summing up the previously stated amounts of emissions from enteric fermentation, manure management and agricultural soils.

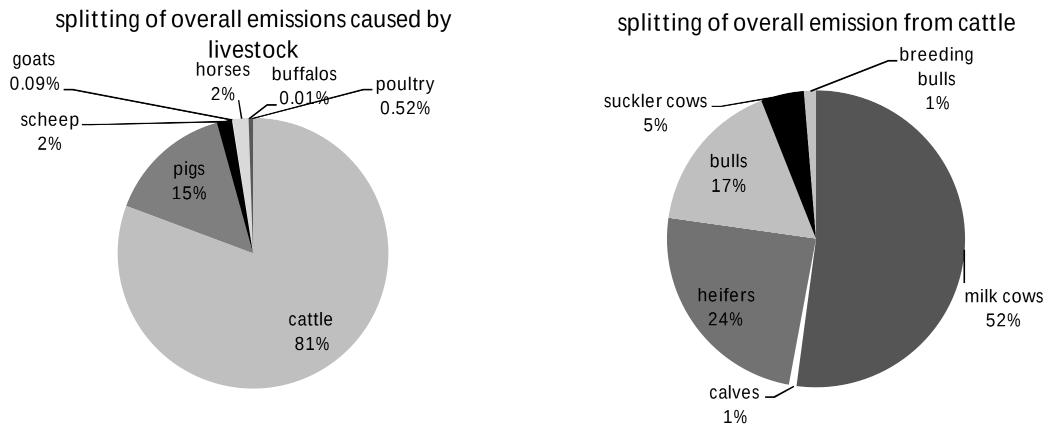
Table 4: **GHGs allocated by emitting groups**

	CO_2	CH_4	N_2O
total amount [t/a]	2,703,000	1,040,118	95,492
dairy cows		43.9%	3.0%
cows+heifers+calves		63.8%	5.3%
cattle total		83.7%	6.1%
pigs		15.0%	1.5%
poultry		0.4%	0.2%
sheep		2.0%	0.0%
other		1.8%	0.3%
soil activities	100.0%	-2.9%	91.9%

Source: own calculation and illustration following DÄMMGEN (2009).

It can be seen that, for CH_4 as well as N_2O, cattle husbandry systems are responsible for the majority of emissions from animal production. Dairy production systems (including cows, heifers and calves) are responsible for the majority of CH_4 (63.8%) and N_2O (5.3%) by themselves. As shown in figure 2, agriculture accounted for about 5.6% of overall 2007 German GHGs (in CO_2-equ.), whereas cattle are responsible for 81% of overall livestock emissions (see the left side of figure 4).

Figure 4: **Fractions of overall emissions from German livestock in CO_2-equivalents assignable to different sources**



Source: own calculations and illustration following DÄMMGEN (2009) and UBA (2009).

Animal husbandry in dairy production systems accounted for nearly 1/3 of Germany's emissions from agricultural production as a whole and about 75% (right diagram of figure 4) of German cattle livestock induced emissions (in CO_2-equivalents) (calculations according to data from DÄMMGEN, 2009).

It can be concluded from the aforementioned summary of emissions from German agriculture and their apportionments to different single responsible sources that dairy production systems as a whole and dairy cows in particular can be identified as the main emitters of agricultural GHG emissions. As highlighted on the left side diagram of figure 4, cattle livestock systems are responsible for 81% of overall GHG emissions from German livestock production. This level produced by cattle husbandry is basically caused by dairy production systems (77%), summing up the percentages due to heifers, calves and milk cows from the right hand side of the diagram. The picture even strengthens by taking soil cultivation into account, which has to be done in light of the 'whole farm' approach, as it builds the basis for fodder production and application of organic and synthetic fertilizers. Hence, dairy production systems will play a decisive role regarding discussions about possible reduction efforts in agricultural systems. Therefore, it is important to do further research on emission abatement strategies, abatement potentials and their economic incentives and effects at the dairy farm level.

2.2 Greenhouse gases in dairy production

In the following sections the most important greenhouse gases (CH_4, N_2O, CO_2) resulting from production processes on dairy farms are defined. The main sources at the farm level important for the occurrence of gaseous emissions are allocated.

2.2.1 Methane

The major sources of agricultural CH_4 emissions (a colorless, odorless gas) are the enteric fermentation of ruminants and their excrements through anaerobic processes (ERG, 2008: p.3; HARTUNG, 2002: p.193). Emissions from enteric fermentation are primarily caused by eructation, stemming mainly from the rumen (87%) and to a small extent from the large intestines (13%) (MURRAY et al., 1976: p.9). It occurs during the process of converting feed material in the animal's fore stomach[9] by the inclusion of different types of microbial species (bacteria, protozoa and fungi) (SHIH et al., 2006: p.4; UBA, 2010: p.365). The intermediate products of these microbial species are converted to CH_4 by methanogenic bacteria[10]. (MOSS et al., 2000: pp.236-237) Concerning differences in livestock type, age, size, fodder intake and fodder digestibility, CH_4 emissions between individual animals and different livestock types vary significantly (CHADWICK and JARVIS, 2004: p.69; FLACHOWSKY and BRADE, 2007: pp.436-438; WILKERSON et al., 1995: p.2403). Furthermore, the lactation periods and proficiency levels of animals have a meaningful impact (JUNGBLUTH et al., 2001: p.135).

But as mentioned before, CH_4 is also emitted from the excrements of the animals. Following e.g. HUSTED (1994) and HARTUNG (2002: p.195), stable floor conditions and manure storage techniques (outdoors or in sub-floor pits) are important CH_4 sources on farms, with different emission rates dependent on the specific type of manure handling (slurry, solid, deep litter and manure removal frequencies from stables). As methane is produced by anaerobic digestion[11] of organic components in the manure, CH_4 emissions are high when liquid storage techniques are used (SHIH et al., 2006: p.4) compared to storage techniques with higher aeration rates (e.g. straw based systems). (CHIANESE et al., 2009b; JANZEN et al., 2006) Animal type and number, temperature, manure amount and management system can thus be named as the primal factors that affect methane emissions. Regarding this, SMITH et al.[12] (2008: p.797) summarized the biophysical reduction potentials of dairy cows. For Western Europe, a reduction potential of 18% is possible by improving feeding strategies. According to these authors, dietary additives can lead to an

[9] A detailed description of ruminal processes is given in BATES (2001: pp.36-37) and MOSS et al. (2000).

[10] Detailed characterization of methanogens in BOADI et al. (2004: p.321).

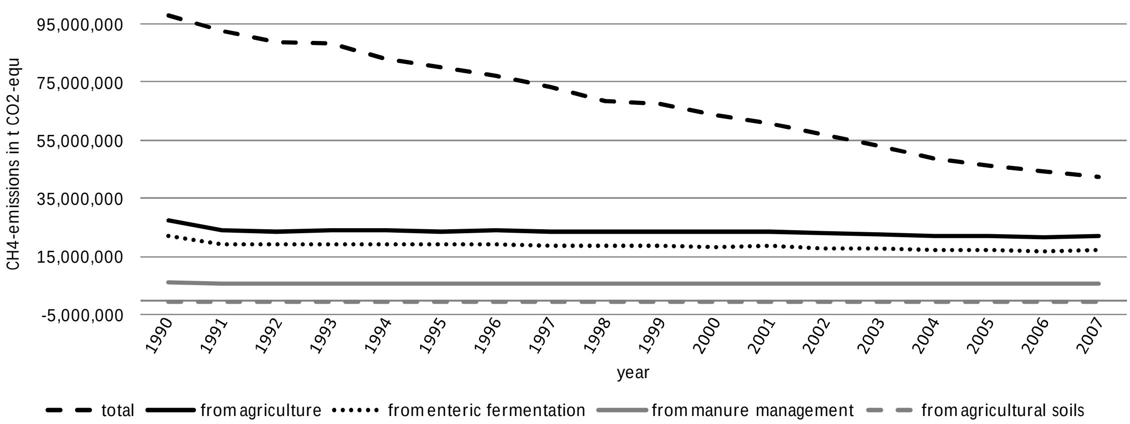
[11] Methane is the end product of the chemical reduction of carbon compounds under anaerobic conditions (CLEMENS et al., 2002: p.203; BURTON and TURNER, 2003: p.89).

[12] The values derived by SMITH at al. (2008) have been adjusted for the non-additivity of the individual options. In reality, interactive effects between practices can occur.

8% reduction and structural management strategies in combination with breeding activities can lead to a 4% decline in methane production. Hence, biophysical activities and implementations in dairy production systems play significant roles regarding methane production at the farm level.

Overall, German methane emissions declined by 56% from 1990 to 2007, as shown in figure 5. But the total amount stemming from agriculture stayed nearly constant, at 21,842,484 t CO_2-equ. Of agricultural emissions in 2007, 76% stemmed from enteric fermentation and 24% came from manure management processes. (DÄMMGEN, 2009) Hence, looking at the development of methane emissions resulting from manure management, agricultural soils and enteric fermentation, abatement efforts in methane emissions failed to have major impacts.

Figure 5: **Methane emissions in t CO_2-equ. from 1990 to 2007 in all of Germany and separated by area of agricultural production**



Source: own illustration following DÄMMGEN (2009) and UBA (2009).

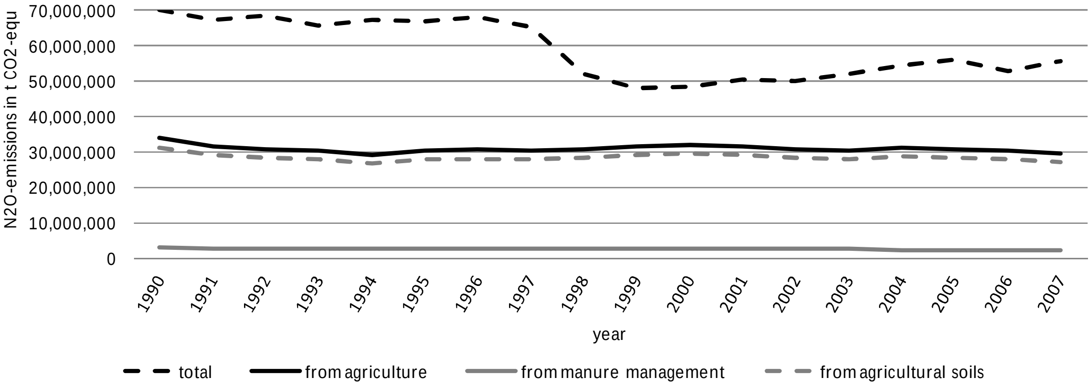
2.2.2 Nitrous oxide

N_2O emissions are mainly related to microbial nitrogen transformation processes in soils and in manure, which are controlled by manure management and application as well as application of synthetic fertilizers. Following FRENEY (1997), soils are meant to be the most important natural source of N_2O production, being responsible for 90% of overall agricultural N_2O emissions. As nitrous oxide emissions originate out of nitrogen, controlling the nitrogen content in the manure and adjusting nitrogen applications to demand are the most readily controllable. As nitrous oxide stems from imperfect denitrification[13] and nitrification[14] of nitrogen in agricultural production processes, all

[13] Formation of nitrogen gas from nitrate reduction (anaerobic) (MONTENY et al., 2006: p.165).

factors influencing rates of imperfect nitrogen conversion are important for controlling N_2O production rates (CHADWICK and JARVIS, 2004: p.70). Nitrous oxide is an intermediate product of an aerobic process, where the availability of oxygen is too low for an optimal nitrification process (SIBBESEN and LIND, 1993). These processes affecting N_2O rates are not only relevant for agricultural soils, but also occur with different surface types of stacked or stored manure, in dung storage areas or on differently constituted stable floors (ROTZ et al., 2010: p.1271). When there is no oxygen available at all, for example in liquid manure without a layer of scum, the potential for N_2O outgassing is relatively low (CLEMENS et al., 2002: p.204). Nevertheless there are several environmental factors affecting nitrous oxide emissions that are not controllable by the farm, for example temperature, dry matter content and soil conditions. A detailed description of the influence of these factors on the rate of N_2O emissions is given e.g. by HARTUNG (2002: pp.193-194). Other factors regarding manure and fertilizer management can be actively controlled (e.g. fitting available N to plant requirements, application technique, rates and times, manure type, stocking densities and type of production system). BOUWMAN (1996), for example, appraised the application of N fertilizer as the main factor in nitrous oxide emission rates from agricultural soils. BROWN et al. (2001: p.1448) also defined fertilizer, animal manure application and urine deposition by grazing livestock as major sources. The housing system also plays a significant role, just as floor conditions and ventilation rates are also important influencing factors.

Figure 6: **Nitrous oxide emissions in t CO_2-equ. from 1990 to 2007 in all of Germany, and separated by sector of agricultural production**



Source: own illustration following DÄMMGEN (2009) and UBA (2009).

[14] Transformation of ammonium to nitrate (aerobic) (MONTENY et al., 2006: p.165); for a detailed description of nitrification and denitrification see BURTON and TURNER (2003: pp.58-64), AMON (1998: p.14).

Total nitrous oxide emissions from Germany decreased significantly until 2007. But emissions from agriculture (sum of N_2O from manure management and soils) stayed nearly constant over time, subject only to small variations (figure 6).

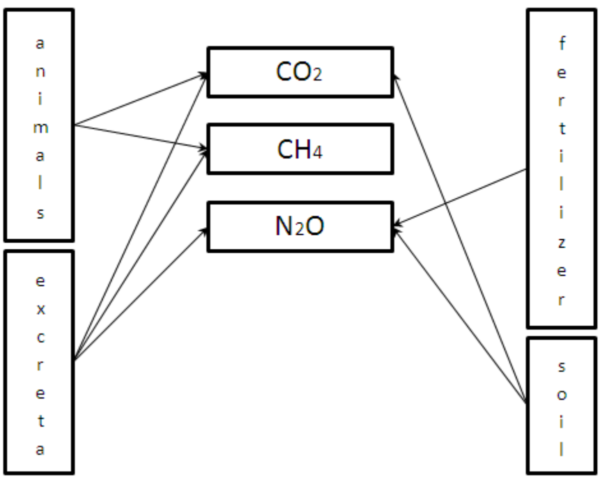
2.2.3 Carbon dioxide

Multiple processes assimilate and emit CO_2 from dairy farms. Cropland activities assimilate carbon dioxide from the atmosphere through photosynthetic activity during crop growth, and emit CO_2 through plant and soil respiration and manure decomposition. As described by CHIANESE et al. (2009a), croplands act as a net sink over a full year, meaning that plants assimilate more carbon dioxide as biomass than they emit during crop growth. In contrast, it is possible that agricultural soil is a net source when permanent grassland is ploughed, which leads to large gaseous emissions of CO_2. So the CO_2 balance between removal by and emissions from agricultural soil is not certain, as also shown in a study by the EPA (2006). In addition, soil emissions and animal respiration are major sources of CO_2 on dairy farms, followed by less meaningful emissions of CO_2 from manure storage systems and barn flooring (BURTON and TURNER, 2003: p.89). (ROTZ et al., 2010: p.1266). Furthermore, general tillage techniques impact carbon sequestration rates.

2.3 Main sources of GHGs

As mentioned in the preceding paragraphs, four sources of GHG production in dairy farming can be identified. Animals as production units by ruminant fermentation, processing of the accumulated excreta, emissions from fertilizer practice and GHGs from soil cultivation are accountable for CO_2, CH_4 and N_2O emissions as visualized in the following figure.

Figure 7: **Main sources of GHGs in dairy production**



Source: own illustration following HARTUNG and PHILLIPS (1994: p.174), LEIP et al. (2010: p.44).

The main sources of CO_2 in dairy production are soil cultivation (land use change, afforestation, change of tillage techniques) followed by animal respiration and excreta (compared to methane and nitrous oxide, CO_2 only represents a very small amount of overall agricultural GHGs). Methane is primarily caused by enteric fermentation processes and manure management. For example, methane emissions from enteric fermentation and manure management accounted for 2/3 of the overall global methane emissions from agriculture in 2005 (EPA, 2005: pp.54-65). High nitrous oxide emission rates are principally generated by sources of high nitrate compounds or activities with large influences on their application, such as animal excreta and fertilizer use in combination with land cultivation.

2.4 Global warming potential

The above-mentioned greenhouse gases from agricultural dairy production do not have exactly the same origins in the production process; gaseous emission rates of CH_4, N_2O and CO_2 have different abetting factors. Circumstances that boost the emission rates of the different gases can even compete against each other, which means e.g., conditions lowering CH_4 emissions lead to higher N_2O emission rates and vice versa. This is caused by the different milieus in which the responsible chemical processes of gaseous emissions proceed. CH_4 emissions are normally supported by anaerobic circumstances, whereas nitrous oxide gases out in higher rates in an aerobic milieu (MONTENEY et al., 2001: p.130; WWF, 2007: p16). Focusing on this, trade-off effects can occur between the levels of emissions of different GHGs with changing production processes and conditions on the farm.

Furthermore, the different GHGs have unequal global warming potentials (GWP) (on the basis of 100 years global warming potential), and so can be expressed in CO_2-equivalents (CO_2-equ.) to obtain a uniform quantifying parameter. As denoted in the table below, different levels of GWP can intensify the aforementioned trade-offs between the gases. For example, changing a production process that lowers, on the one hand, emissions of CO_2 by one kilogram but on the other hand increases the emission of N_2O by one kilogram would lead to an overall increase in emissions of 309 kg CO_2-equ. The reason for this is that the GWP of one kg of nitrous oxide is 310 times that of one kg of carbon dioxide. The results of such trade-offs between different types of GHGs can be derived from the following table.

Table 5: **Global warming potentials of the different greenhouse gases expressed in CO_2-equivalents per unit of the specific gas**

Greenhouse gas	composite	GWP (in CO_2-equ.)
Carbon Dioxide	CO_2	1
Methane	CH_4	21
Nitrous Oxide	N_2O	310

Source: own illustration following UBA (2009: p.57).

In terms of potential mitigation possibilities at the farm level by changing production processes or restructuring the overall production portfolio, the abatement effectiveness of the mitigation options always has to be evaluated by considering also their impacts on the accruement of other GHG types and their relating GWPs.

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Chapter 3: **Up to date relevant GHG abatement options in German agricultural dairy production systems**[15]

Abstract

Mitigation of greenhouse gases is receiving more and more interest in current political discussion and also in the agricultural context. This technical paper comprises a summary and categorization of different applicable greenhouse gas mitigation options discussed in the literature that are able to effectively reduce GHG emissions arising from dairy production facilities in Germany. GHG mitigation options are therefore identified which have great differences in time resolution with regard to their implementation and practical applicability as well as effects on the relevant gases (CH_4, N_2O, CO_2). Further on costs induced by mitigation strategies show different uncertainties concerning sensitivities to farm exogenous impacts.

Keywords: *GHG mitigation, greenhouse gases, GHG abatement options.*

[15] This chapter is based on a technical paper which was developed during the work on the DFG funded project with reference number HO 3780/2-1. The technical paper is available on the project related web-page of the Institute of Food and Resource Economics of the University of Bonn as LENGERS, B. (2012): Up to date relevant GHG abatement options in German agricultural dairy production systems. http://www.ilr.uni-bonn.de/agpo/rsrch/dfg-ghgabat/dfgabat_e.htm

3.1 Categorization of abatement options

The results of this work were obtained during the construction of a farm-level model approach for German dairy farms[16] analyzing GHG mitigation options that have to be implemented in the model. In the following only mitigation options that are applicable up to now and allowed in Germany and the European area are considered and explained.

So far, there are several abatement options for the mitigation of methane (CH_4), carbon dioxide (CO_2) and nitrous oxide (N_2O) emissions that are discussed in the literature on agricultural production and for the dairy sector in particular. SMITH et al. (2008: p.790) classify the mitigation of emissions into reduction (e.g. through better C and N efficiency and fodder optimization), enhancement (accumulating C in soil), and avoidance of emissions (replacing, for example, production requiring high energy with more extensive production). Admittedly, the approaches that best mitigate emissions depend on many conditions and thus vary between regions, farm sizes, and production orientations. So there are great differences concerning the practical feasibility of the mitigation options. From an application-oriented point of view a mitigation option can mean employment or adaption of existing technologies (e.g. change of milk yield potential, fodder composition, or N-fertilizer intensity) or involvement of new technologies (e.g. new type of manure storage, feed additives like oils, new manure application techniques). Furthermore, we want to use a fully dynamic framework as our general modeling approach considers a long planning horizon to be able to consider developments and production decisions that are connected with long term investments. Because of this we also have to consider the time resolution and the impact of abatement options on the possible development paths of the firms. Referring to this, we can classify mitigation possibilities into two general groups, *permanent* and *variable*.

3.1.1 Permanent options

Permanent abatement options are mainly characterized by investment decisions or management changes which influence the production process for a longer period, not allowing for a flexible adjustment over time (e.g. from month to month or from one year to another). So emissions are reduced indefinitely compared with the situation without usage of the permanent option. Options include, for example, coverage of manure storage, choice

[16] Named DAIRYDYN (LENGERS and BRITZ, 2012).

of specific floor conditions in the stable, decisions about the general stable type (slatted floor, deep litter, slurry, etc.) and size, animal-keeping system, or different manure storage and application techniques which are discrete in time but influence possible future decision paths. This is a special attribute of permanent options, leading to path dependencies. The decisions for permanent options are 0-1 decisions (binary variables), whereas the amount of their implementation is of integer character as they can only be implemented as a whole. Once applied, a reversal of the options in the next years would cause sunk costs and perhaps extra costs of uninstalling or replacement of the basic structure of buildings or machines. The mitigation effect of the single options can be fixed as a specific total amount or a percentage reduction compared to the situation without the option. The costs that result due to the specific permanent mitigation options can be easily derived from the investment cost of the single option (costs calculated as depreciation cost over useful lifetime or operating hours).

3.1.2 Variable options

This type of abatement option facilitates a flexible adjustment of the decision maker facing new conditions or constraints. The abatement strategies that are summarized under this category are more reversible or variable than the permanent options. Allowing the farmer more flexible reactions from year to year or from month to month is the main advantage of these options. Also the temporal pattern of influence on the GHGs may be different among mitigation options. Some of the variable options involve new technologies (like fodder additives) but are also highly flexible in usage over time. But the majority of variable options are presented by changes to existing production processes and techniques (which do not contain of additional activities or processes like new types of manure application), implemented endogenously in the existing farm processes by changing the level of process relevant variables. Examples are fodder optimization or improvement of fertilizer use, which can influence the emission of GHGs. Variable options allow for a dynamic adjustment of the farmer’s decision variables which impact, for example, breeding activities, herd size management, fertilizer intensity, fodder optimization processes, and the choice of fodder ingredients. It is obvious that most of these decision variables influencing the flexible mitigation strategies are continuous variables which can be varied in the scope of the side conditions (e.g. minimum dry matter, maximum fodder). Mitigation effects of these process-based options are calculated via emission functions concerning ruminant fermentation or manure-handling processes depending on the flexible decision variables of the farm. The costs of GHG abatement are calculated directly from the implementation cost or indirectly through a change of variables in the dairy production process. When the cost of inputs and the amount of output and revenues change, the economic development of each farm related to the change in calculated GHG emissions will show the mitigation costs caused by the abatement strategies.

3.2 Description of mitigation options

In the following sections, relevant mitigation options for the German or European dairy context are named and described, referring to several up to date works in the literature (e.g. BOADI et al., 2004; FLACHOWSKY and BRADE, 2007; KTBL, 2002; OENEMA et al., 2001; OSTERBURG et al., 2009a, 2009b; PATRA, 2012; SCHILS et al., 2006; VABITSCH et al., 2004: p.197; WEISKE, 2005; WEISKE and MICHEL, 2007). The list of mitigation options might appear relatively restricted and short, and a subject-specific reader may know of additional GHG abatement strategies (like those listed in the SAC-Report by MORAN et al. (2008: p.148) for example). But, as mentioned before, only mitigation options that are practically applicable within the tight system boundaries of the modeled dairy farms are to be considered. Therefore qualitative valuations like those from OSTERBURG et al. (2009a: pp.41) concerning practicability, efficiency, and uncertainty are used. Furthermore, abatement options which are operative following several research results but in conflict with German or European law are excluded from the examination (like for example addition of types of antibiotics to the ration).

3.2.1 Permanent options

3.2.1.1 Stable type

The animal-keeping system in the stable has a significant role concerning emission rates of methane and nitrous oxide as shown for example by an overview of HARTUNG (2002: pp.195–196). Emissions in the stable are determined by floor conditions as well as aeration rates and by whether the stable has a solid or slatted floor, slurry based or deep litter systems, and tied or free stalls (WEISKE and MICHEL, 2007: p.15). Additionally, the question of whether the herd spends 365 days in the stable or if pasture is also available for the animals has to be taken into account. But pasture management is in turn a variable option which can in principle be combined with all of the different stable types as long as grazing areas are available. As shown by the table below, the choice of stable type with its specific floor conditions can have a meaningful impact on the GHG emission amounts. But decisions on stable types are not reversible or flexible so that in an existing enterprise this management or control option will only be considered if replacement or expansion investments take place.

Table 1: **CH_4 and N_2O Emissions from dairy stable systems in grams per livestock unit[17] and day**

System:	CH4	author	N2O	author
dairy in tied stall	327	Kinsman et al., 1995	0.14-1.19	Amon et al., 1998
	120	Groot Koerkamp and Uenk, 1997		
	194	Amon et al., 1998		
dairy in free stall	320	Sneath et al., 1997	0.8	Sneath et al., 1997
	265	Groot Koerkamp and Uenk, 1997	0.3-2.9	Brose, 2000
	267-390	Seipelt, 1999		
	200-250	Brose, 2000		
dairy deep litter	782	Seipelt, 1999	2.01	Amon et al., 1998

Source: own illustration following HARTUNG (2002: pp.195–196).

As shown by the table and stated in CHADWICK and JARVIS (2004: p.72), slurry based systems have lower nitrous oxide emissions from animal housing and storages because of their more anaerobic milieu. That there is little or no emission of nitrous oxide due to slurry-based stable systems is also underlined by the findings of THORMAN et al. (2003), but who found 4–5 mg of N_2O-N/m²/d emissions from cattle housed with straw bedding. Following this, a change from straw-based systems to slurry-based systems can be valued as a meaningful N_2O mitigation option. But, in turn, storing animal excreta under anaerobic conditions in, for example, sub-floor pits can boost CH_4 production from the managed manure. The effect of different floor conditions in the case of solid or slatted floor is not clearly identifiable up to now. For example SCHNEIDER et al. (2005) and ZHANG et al. (2005) state that on average solid floors cause slightly higher methane emissions in free stalls compared with slatted floors. Contrary to that, SNELL et al. (2003) quantified lower CH_4 emissions for solid floor conditions. In the case of N_2O emissions SCHNEIDER et al. measure higher emissions from slatted floors, while ZHANG et al. on the contrary found lower nitrous oxide measurements for slatted floors.[18]

3.2.1.2 Manure storage techniques

CH_4 and N_2O are emitted during the storage of manure and dung outside the stable. The emission rates depend on the total deposition quantity of the animals, storage time, and the

[17] One livestock unit is equivalent to 500 kg live weight.

[18] For a tabulated listing of emission measurements given in the literature see also SCHIEFLER and BÜSCHER (2011: p.158).

additional substances that allow a conversion into GHGs (for example, straw is added to deep litter systems). The feed quality and ingredients, feeding techniques, animal type, and milk output level have a measurable influence on the control of gaseous emissions from manure storages in particular. (BLOK and DEJAGER, 1994: pp.27-28; CLEMENS et al., 2002: p.203) But these factors belong to mitigation techniques based on feeding and intensity management, which are discussed in later sections. As an external factor which is not controllable from the farm side, environmental aspects such as temperature and condensation rates are also important. With a proper choice of storage techniques one has the possibility to react to given environmental location factors as well as to manure attributes induced by feeding practices.

For liquid manure storage, we can assume two different types, lagoons and silos (surface liquid manure reservoir), among which the silo variant can be identified as the dominant one for the German context and general gaseous activities in both systems can be assumed to be equal. Concerning the surface of the liquid manure, different possibilities exist. Manure without a layer of scum is anaerobic (except for the upper 1–1.2 cm), so the potential for nitrous oxide emissions is near zero (CLEMENS et al., 2002: p.204). But on the other hand this is the perfect condition for CH_4 accumulation. Especially for cattle slurry, the formation of a layer of scum is possible because of the high content of organic matter, which increases the risk of N_2O emissions. Additionally, artificial coverage options exist for liquid manure, like finely chopped straw. This option is mainly adopted to reduce ammonium but it can yield N_2O emissions because of the aerobic processes in the covering layer. Conflicting statements exist in the literature concerning the effect of straw coverage on CH_4 emission rates. (CLEMENS et al., 2002: p.204) SOMMER et al. (2000) state that methane emissions would diminish by 38% on average. Contrary to this, HARTUNG and MONTENEY (2000) and WULF et al. (2002: p.488) mention an increase in CH_4 quantities, which could be explained by the addition of carbon to the slurry. Straw cover as well as slurry aeration is also identified as having a negative overall environmental effect by the analysis of AMON et al. (2004: p.95). As an additional and most effective control technique, foil coverage is mentioned, which enables a totally airproofed closure of the storage to reduce CH_4 and N_2O emissions. Technical degrees of efficiency of between 80% and 100% can be assumed for this mitigation option (KRENTLER, 1999, cited in OSTERBURG et al., 2009a: p.64). Further possibilities exist with regard to the handling of liquid slurry. Separation of manure, for example, avoids the production of N_2O from the liquid compound because a layer of scum cannot build up (CLEMENS et al., 2002: p.205; OSTERBURG et al., 2009a: p.64).

Storage techniques for dung do not show such variety. Following CLEMENS et al. (2002: p.208) this type of animal excretion emits mainly N_2O. The rate of aeration of stored dung seems to be very relevant for the emission amounts, whereas N_2O and CH_4 emissions can be diminished significantly through largely aerobic storage, which means a high technical input is required for loosening, comminuting, and moving the material.

The costs that are connected with different storage techniques arise mainly from the building and installation costs. Partly, additional requirements for machinery or labor input can arise through such GHG mitigation options.

3.2.1.3 Application technique

With regard to the amount of GHG emission that is controllable by application techniques, in general the exogenous factors that influence the processes are the same as those that affect the storage techniques. Temperature, soil attributes, general consistence and N content of the manure that has to be recovered are important and exogenously given or not controllable by the choice of application technique. Further on, application quantity and time markedly impact the occurrence of GHGs. (CHADWICK et al., 2011) The application technique is relevant not only for the liquid manure or dung that is produced by the dairy farm but also for the handling of fertilizers. Following CLEMENS et al. (2002: pp.210–212), broad spreading, trail hose, and injection are the most relevant application techniques for liquid manure. But with regard to research results of GHG emission impacts of the different slurry application techniques, CLEMENS et al. (2002: p.211) mention great differences in the emission effects that occur and DITTERT et al. (2001) underline that injection can result in increasing N_2O emissions. However, there are also authors who quantify the mitigation potential of application techniques as to be negligible (OSTERBURG et al., 2009a: p.72). Furthermore, there is also a possibility to implement different application techniques as flexible options if contract work is allowed. (The option of adapting a proper application technique for manure can in certain cases also belong to the variable or temporal options if for example the extraction of organic as well as synthetic fertilizers is handled by external enterprises.) This is normally more expensive per kilogram of fertilizer applied in comparison to private mechanization (calculated over the full depreciation time and high capacity utilization), but leads to a higher flexibility and less sunk cost in the case of changes in application type over short intervals.

3.2.2 Variable options

3.2.2.1 Fodder optimization

Dairy production leads to specific requirement functions for forage and feed with regard to lactating cows, heifers, and calves. These requirements have to be fulfilled by the fodder composition. Improving the quality and nutrition significantly reduces the methane production per animal (holding output constant) and per unit of product. The optimization of fodder ingredients and feed composition enhances the animal performance and means that a higher fraction of the fodder energy is converted to useful sinks (e.g. weight gain, milk output, productive lifetime, body condition). Furthermore the digestibility of fodder ingredients plays a significant role as higher digestibility lowers methane production, so for a low-emitting fodder composition only compounds with high digestibility rates are relevant (HELLEBRAND and MUNACK, 1995). (O´MARA, 2004) This optimization of the feed conversion efficiency can lead to substitutions of roughage with concentrates (BATES, 2001: p.42). Overall it can be assumed that the minimum or maximum requirements of the animals arising from maintenance and activity should be fulfilled optimally so that no energy or nutrient overhang exists which would lead to higher emissions of N_2O or CH_4 from the manure. Additionally the feed conversion efficiency is very important. As also mentioned by BOADI et al. (2004: p.323) and JOHNSON et al. (1996) the fermentation of cell wall carbohydrates influences methane production. Concerning the energy source of concentrates less CH_4 is emitted by starch in the ration compared to the fermentation of soluble sugars, so substituting sugar with starch in the ration (concentrate) is meant to diminish methane emissions significantly (JOHNSON et al., 1996; MILLS et al., 2001: pp.1591–1593). The main factors of the fodder ingredients which influence the occurrence of emissions from enteric fermentation are also described precisely in MONTENY et al. (2006: p.164), for example.

3.2.2.2 Breeding activities/replacement rates

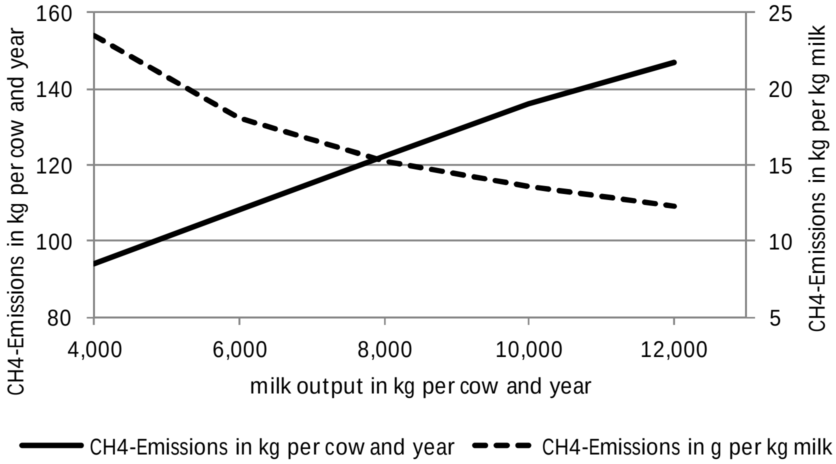
“The most promising approach for reducing methane emissions from livestock is by improving the productivity and efficiency of livestock production, through better nutrition and genetics” (ERG, 2008: p.4). Herds consist of animals with different genetic potential. With selective breeding activities the farmer can influence the overall milk yield potential of the farm’s livestock or even breed low-emitting phenotypes. Raising the milk output per cow through more sharp selection decreases the amount of methane emitted per kilogram of milk because emissions that arise from energy requirements for maintenance are spread over a larger output (CAPPER et al., 2009: pp.2163–2165; ERG, 2008: p.4; MONTENY et al., 2006: p.165). Conditional upon the increasing milk yield potential of the cows, also emissions per cow are typically higher because of higher feed intake. But as the productivity of each cow increases, the farm manager can reduce the herd size to produce the same amount of output (ERG, 2008: p.8). This aspect is also discussed by FLACHOWSKY and BRADE (2007: p.440) and CORRÉ and OENEMA (1998), who name the increase of yield level as the most efficient option to abate methane emissions from ruminant fermentation so far. In addition to the genetic characteristics, breeding and selection activities can yield a higher longevity of the animals, which will decrease methane emissions from the herd (O´MARA, 2004). The longer each lactating animal stays in the herd, the fewer calves and heifers are needed for replacement and reproduction, which results in lower total methane emissions because of a lower total amount of livestock. Thus controlling the useful lifetime and hence the number of lactations per cow has a meaningful impact on methane emissions (FLACHOWSKY and BRADE, 2007: pp.441–442). But generally one can say that there exists a trade-off between breeding for higher milk yield and increasing the longevity of the animals, because a higher proficiency level of cows is attended by a reduction of their expected useful lifetime. In addition, the output curve of each cow influences her economically efficient useful lifetime and thus the replacement rates. The decline of the milk output of each cow in later lactation phases also affects the emissions that occur per kilogram of output. So, beside an economically efficient useful lifetime, an emission-efficient number of lactation periods can also be derived for each cow. Hence, changes of the replacement rate are an applicable management-oriented abatement strategy in dairy production (WEISKE et al., 2004: p.140).

3.2.2.3 Intensity management

Increasing the production intensity means that the output per production unit is raised. For arable production this means that input levels of fertilizer and pesticides will be heightened and optimized to reach a higher yield level per hectare. The more intensive animal production is to raise output level per animal and per stable place, the higher is the input level of nutrients, labor, investments, and perhaps veterinary costs. A higher intensity level, through for example higher rates of concentrate, normally causes higher emissions of N_2O and CH_4, overall and per production unit. But depending on the relation between the emissions that originate from nutrient inputs, fertilizer, or manure storage/application and the production output quantities, intensity management can be an option to diminish emission quantities per unit of product (BATES, 2001: p.39; OSTERBURG et al., 2009a:

p.54). So, increasing productivity decreases emissions per product unit (kilogram of milk) (GERBER et al., 2011, JOHNSON et al., 1996; PATRA, 2012: p.1932). This effect of CH_4 reduction per kilogram of milk with increasing intensity level can be illustrated by the research results of FLACHOWSKY and BRADE, shown in the following figure.

Figure 1: **CH_4-emissions by fermentation process in the rumen**

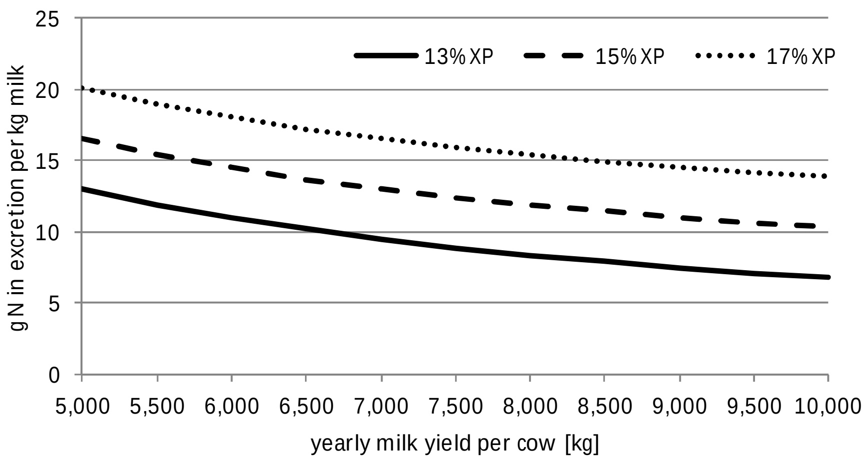


Source: own illustration following FLACHOWSKY and BRADE (2007: p.438).

To reach higher milk yields, the fodder intake has to be increased. With higher milk output levels, nutrient requirements also go up. But as the dry matter intake (DMI) capacity of cows is restricted, the energy concentration per kilogram of DMI has to be increased. This can normally only be achieved with higher concentrate rates in the fodder, which leads to a higher DMI intake per day and a higher level of milk yield. As shown by the figure above, this means an increase in CH_4 emissions per cow but diminishing emissions per kilogram of product. (LOVETT et al., 2005) This can for example be explained by the results of MILLS et al. (2001: pp.1590–1591) and JOHNSON and JOHNSON (1995), who determine that increasing the DMI of cows diminishes the proportion of gross energy lost as CH_4. So the concentrate to roughage ratio has to be aligned with the maximum DMI to provide the energy which is necessary to achieve a desired level of milk yield. Also, KIRCHGESSNER et al. (1995) show that an increase of milk output quantity per cow from 5,000 to 10,000 kg per year increases the methane production per animal by 23% but reduces CH_4 production per kilogram of milk by 40%. As already mentioned in relation to the option of breeding for higher genetic milk yield potential, increasing single-animal efficiency also reduces GHG emissions per animal through optimal intensity management. A drawback of this option is that an increase in intensity level to fully utilize the milk yield potential of the cows will lead to a decline of their useful lifetime, with fewer lactation periods, and hence animal replacement rate increases, which leads to a higher demand for calves and heifers (a bigger animal population means more GHG emissions). But, following FLACHOWSKY and BRADE (2007: p.440), increasing the intensity level and thus the output per production unit can be named as the most effective option for mitigating CH_4 emissions so far.

Increasing the milk yield can also impact the gaseous emissions of nitrous oxide, as nitrate content in the manure is a source of N_2O emissions and higher milk yield levels diminish the N content in excreta per kilogram of milk output (KIRCHGESSNER et al., 1991a). This is described in CLEMENS and AHLGRIMM (2001: p.291) using N-excreta functions from KIRCHGESSNER et al. (1993). This circumstance can be a result of asymptotically diminishing excreta amounts per kilogram of milk output with increasing milk yield level as shown by WINDISCH et al. (1991). These authors derived interesting linear relationships between feed intake, milk yield level, and resulting excreta amounts, where the relationship between daily feed intake in kilogram of dry matter and the amount of manure occurring is calculated as 2.4 + 2.88 * (kg DMI/day) = (kg manure). The formula depending on daily milk output was derived as 2.6 + 0.92 * (kg milk/day) = (kg manure). (WINDISCH et al., 1991) The factor of 0.92 shows an underproportional increase in excreta with increasing milk output level which supports the asymptotical decline in excreta amounts per kilogram of milk output as mentioned before.

Figure 2: **Nitrate content in excreta per kilogram of milk output** (XP = crude protein content in fodder)



Source: own calculation and illustration following KIRCHGESSNER et al. (1993).

Following the information of the above figure, keeping the overall milk production of a farm constant and increasing milk output per cow increases the milk produced per units nitrogen (N) in excreta and thus lowers the potential of N_2O emissions per kilogram of milk. Furthermore, controlling the crude protein content (XP) of the fodder composition influences the N excretion rates significantly. For example the N excretion per kilogram of milk of a cow with yearly milk output of 7,000 kg increases by 80% when the XP content is increased from 13% to 17%.

This intensification effect on the emissions per kilogram of product is also true for arable production. By increasing yield levels per hectare, N_2O background emissions occurring from soils are allocated to more output and hence overall nitrous oxide emissions per product unit will diminish with increasing per hectare yields. So, for example optimizing fertilizer use to reach higher yield levels is named as an effective mitigation option by BURNEY et al. (2010).

3.2.2.4 N-reduced feeding

The reduction of protein in the fodder, which is tailored to requirements of the feedstock, reduces the amount of N excreted. This causes lower GHG emissions (N_2O) from manure in the stable and from storage and N_2O emissions stemming from manure application. (OSTERBURG et al., 2009a: p.50) This requires an N-adjusted feeding strategy in line with the animal requirements because nutrition combined with animal performance affects the N excretion (WEISKE and MICHEL, 2007: p.9). Hence, improving N use efficiency and reducing overall N input into the system is a meaningful measure to reduce N_2O emissions as well as the overall risk of N losses (AARTS et al., 2000).

3.2.2.5 Fertilizer practice

A reduction of exogenous mineral and organic fertilizer (e.g. ammonia and urea) purchased by farms in arable production leads to lower N_2O emissions. Fertilizer management is an important regulating tool where oversupply of nitrate from fertilizer is to be avoided. This means a fertilizer adjustment is to be applied in line with demand, and thus the application amount and time have to fit the soil conditions and the yield-related requirement of the plant growth. The lower the oversupply of fertilizer, the lower is the emission potential. A further important point to mention is that fertilizer practice has to be harmonized with the application of farm fertilizer from animal production because both provide nitrogen. (OSTERBURG et al., 2009a: p.56) CH_4 emissions are not affected by N fertilizer practice (VELTHOF et al., 2002: p.506).

3.2.2.6 Soil cultivation

Generally one has to differentiate between arable farm land and continuous grassland (mostly pasture). With regard to these two general types of soil usages, the issue of whether or not ploughing up of continuous grassland is allowed is important because when pasture is ploughed fractions of soil carbon are transformed into CO_2. In the process, nitrate is also released as N_2O. Hence, avoiding a change in use of continuous pasture abates potential CO_2 and N_2O emissions. (OSTERBURG et al., 2009a: p.44) Consequentially this is a kind of general assumption with regard to whether or not ploughing of continuous pasture is admissible. For cultivation of arable farmland it is also possible to change the method of husbandry. Reduction of soil cultivation, addition of organic matter, and change of crop rotation are possibilities to increase the carbon content of the soil (OSTERBURG et al., 2009a: p.58). Further on, deposition of CH_4 is possible for grass and arable land but differs with regard to level (DÄMMGEN, 2009: p.315).

The costs of these management strategies are composed of operating costs and different influences on market revenues for the yields obtained as well as on production costs due to substitution of fodder components with, for example, concentrates.

3.2.2.7 Herd size management/crop-growing decisions in farming

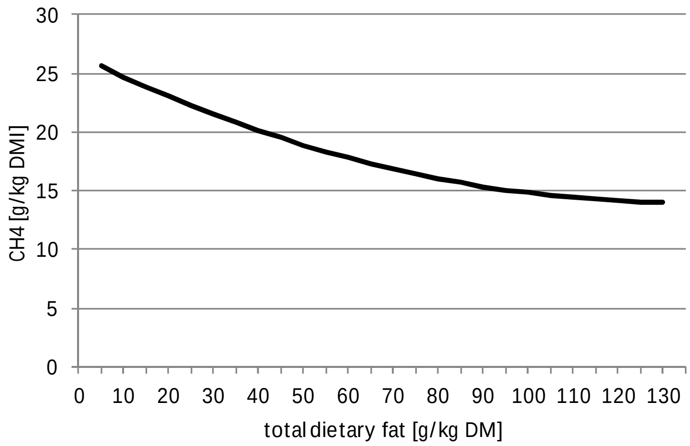
This management option is on the one hand the simplest emission mitigation option but on the other hand most likely the most expensive. By lowering the herd size the farmer decreases the possible number of emitters on the farm. But concerning the mitigation cost it is obvious that with the removal of a single cow the farm loses the gross margin of this lactating cow. So this option is supposed to be an expensive way of mitigating GHG gases. Similarly a reduction of one hectare of a cash crop mitigates the emissions that are produced by the crop growing but also results in the loss of its gross margin.

3.2.2.8 Feed additives/fat content

There are different types of feed additives mentioned in the literature. The Eastern Research Group (ERG, 2008: pp.6–7) describes the mitigation effect of feed supplementation with fats, oils, propionate precursors, and secondary metabolites. These applications influence the metabolic fermentation processes and the methane emissions that occur. According to MURPHY et al. (1995) and ASHES et al. (1997) the addition of fats and oils leads to a higher energy density of diets, raises the milk yield, and enriches the fat content of the milk. Furthermore research results show that fats and oils reduce CH_4

production (DONG et al., 1997; MACHMÜLLER and KREUZER, 1999; JORDAN et al., 2004: pp.124–130) but are also meant to have only a short term influence because of the adaption of methanogenic bacteria in the rumen to the new fodder components (ERG, 2008: p.6). GRAINGER and BEAUCHEMIN (2011) derived a diminishing effect of higher dietary fat content on the enteric methane production per kilogram of dry matter intake, visualized in figure 3. But the results of previous studies only stem from short term experiments and DUGMORE (2005) mentions that the feed content of fats and oils should not exceed 8% of feed dry matter. But nevertheless, digestibility is affected by higher fat contents.

Figure 3: **Methane emissions per kilogram of dry matter intake depending on dietary fat content**



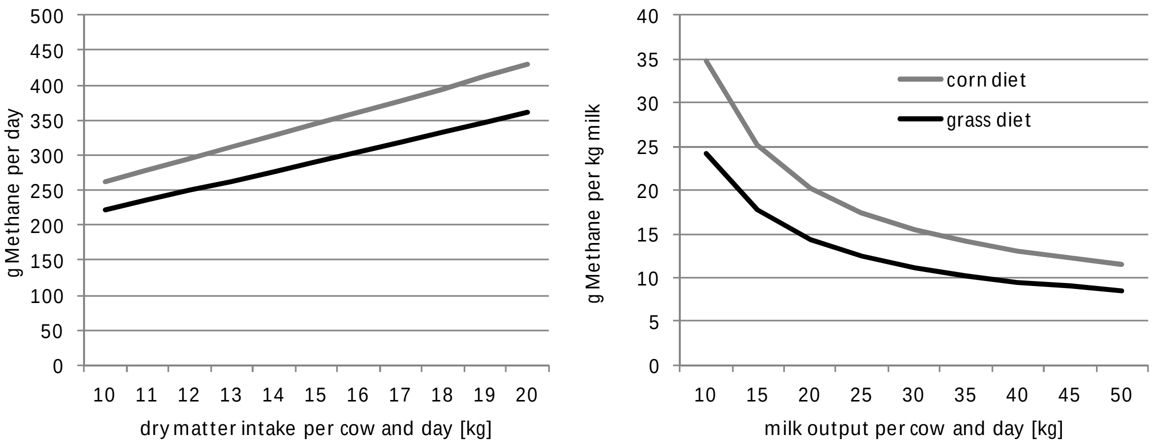
Source: own calculation and illustration following GRAINGER and BEAUCHEMIN (2011: p.310).

Propionate precursors convey the production of propionate instead of methane from the transformation process of hydrogen in the rumen (O´MARA, 2004). But according to O´MARA the precursors are currently very expensive and their effects are not yet quantifiable in sufficient detail to be implemented into the model approach of this work. Secondary metabolites like saponins and tannins are other options for methane reduction from metabolic fermentation through inhibiting the corrosion of organic components (PEN et al., 2006). Furthermore the addition of ionophores, a kind of antibiotic, is also discussed in the literature (BOADI et al., 2004: p.326). VANNEVEL and DEMEYER (1996) discuss the potential of methanogenetic control by feed additives like polyhalogenated compounds, other antibiotics, lipids and probiotics and result that possible reduction effects are not long lasting and that negative effects on digestion and overall productivity may occur by toxicological effects. Hence, as the addition of antibiotics as fodder components has been banned by European law since 2006 [Enactment (EG) No. 1831/2003] and the mitigation effects are not long-lasting according to JOHNSON and JOHNSON (1995) as well as VANNEVEL and DEMEYER (1996), these options are not of relevance for this work.

3.2.2.9 Pasture management/increased grazing

Adequate management of grazing can lead to lower overall emissions by the herd. The addition of pasture in the ration can diminish the gas fluxes of CH_4 per kilogram of DMI and per kilogram of raw product as shown in the following figure.

Figure 4: **Methane emissions by dairy cows with a maize- and herbage-based feeding strategy**



Source: own calculation and illustration following KIRCHGESSNER et al. (1991b: pp.93–96).

The rotation rate among pastures influences the availability of higher quality pasture for the animals, which directly impacts the productivity and decreases methane emissions per unit of product from enteric fermentation (BOADI et al., 2004: p.325; MCCAUGHEY et al., 1997). The analysis performed by MCCAUGHEY et al. also demonstrated that there is a significant difference in CH_4 emission rates depending on whether pastures are rotationally or permanently grazed, and a 9% lower methane production per hectare per day was measured on rotationally used pastures compared to continuous grazing. Furthermore the use of pasture diminishes emissions from manure storage (because there is less manure in storage systems) but increases N_2O emissions from deposition by grazing animals (CHADWICK and JARVIS, 2004: p.72). Comparable results are derived by VELTHOF et al. (2002) using the MITERRA-DSS model, showing that with a higher grazing portion N_2O also increases but methane emissions decrease. But also in the case of fertilizer practice on croplands, tailoring nutrient additions to pasture plant uptake reduces N_2O emissions (FOLLETT et al., 2001). Furthermore, DURU et al. (2007: p.208) mention that an increase in N-use efficiency by controlling the grazing rates and times to guarantee a balanced herbage growth to N ratio in line with feed intake by grazing animals helps to reduce the risk of N losses. But it is important that sufficient pasture areas are available near to the farm to allow an optimal and adequate inclusion of pasture strategies in the feeding plan of the farm (OSTERBURG et al., 2009a: p.53). This circumstance is primary dependent on the geographic region in which specific dairy farms are located. In 2009, about 40% of the German dairy livestock had access to pasture, with slight variations depending on farm herd sizes (STATISTISCHES BUNDESAMT, 2010). Hence, it is an important abatement possibility that is appropriate for a large part of the German dairy livestock population. As for normal in-stable feeding, a higher pasture rate also requires an adapted addition of concentrate to regulate the milk yield level to make the best use of the CH_4 mitigation potential (FLACHOWSKY and BRADE, 2007: p.444).

3.3 Impact of mitigation options

Summing up the above stated influences of the abatement options on emissions of CO_2, N_2O and CH_4, the following table lists the described GHG mitigation options and illustrates which gases can be influenced by the single abatement strategies; these findings are comparable with results of SMITH et al. (2008: p.791).

Table 2: **Influence of abatement options on different GHGs**

	CH_4	N_2O	CO_2
permanent			
stable type	x	x	
manure management techniques	x	x	
application techniques		x	
temporal			
fodder optimization	x	x	
breeding activities	x		
intensity management	x	x	
N-reduced feeding		x	
fertilizer practice		x	
soil cultivation		x	x
herd size managment, crop growing decisions	x	x	x
feed additives/ fat content	x		
pasture management/ increase grazing	x	x	x

Source: own illustration.

As visualized by table 2, many of the different abatement strategies affect the emission rates of not just one specific gas but two or even all three of the relevant GHGs. Attention to this matter is important because some options can cause the emission rates of different gases to develop in opposite directions (ROBERTSON and GRACE, 2004: p.61; SCHNEIDER and MCCARL, 2005: p.9). If for example the manure management is changed towards low emission rates of methane through building up aerobic conditions in manure storage, this change will boost the outgassing of nitrous oxide, which has a much higher global warming potential (21 for CH_4 and 310 for N_2O). This fact underlines the importance of paying attention to these interactions and trade-offs between different management strategies and emission types.

Furthermore, the aim of the underlying model approach is to simulate mitigation activities of a great variety of different dairy farms. Regarding this, it is important to underline that mitigation practices that do a good job of reducing GHGs on one specific dairy farm type may be less effective when farm characteristics are different (SMITH et al., 2008: p.798). So far, there are several studies dealing inter alia with the level of mitigation potential of different abatement options (e.g. SMITH et al., 2008: p.802). These allow for a relative ordering (ordinal scale) of the mitigation options with respect to their abatement capacities. Obviously, here the abatement potential is evaluated from an engineering point of view. The inclusion of more complex economic aspects in addition may lead to totally different results by also recognizing farm-dependent cost efficiencies of different mitigation strategies. Thus, it is very important to build up a highly disaggregated and detailed model to incorporate abatement effectiveness and efficiency of mitigation strategies, when their effectiveness and efficiency depend on the specific structural farm conditions[19]. Therefore it is not possible to offer a universal list of mitigation options with given mitigation factors and costs to all farms; the proposed practices have to be calculated for each individual agricultural production system on the farm.

3.4 Sensitivity of different mitigation options

As mentioned before, the main aim of this work is to derive cost functions in dairy production which occur through GHG-abating adjustments of the production process on individual farms. Because of this, the costs associated with the use of each single option are very important for the derivation of abatement and marginal abatement costs for GHGs. But, as several mitigation options like milk yield intensity management or fertilizer practice have an impact on production levels, the mitigation costs that result through diminishing production amounts are directly linked to the market price of milk and cash

[19] We have done this by offering a highly detailed resolution in the model DAIRYDYN. Each defined farm is confronted with a highly flexible set of abatement options as not only the options themselves, but also the level of their application can be chosen flexibly if they are coupled to continuous variables in the model approach. The optimal choice for a specific farm is hence defined by the interplay of the chosen GHG indicator, the abatement potential and the economic effect of the strategy.

crops (e.g. slaughtering of a cow as a GHG-mitigation option becomes more expensive when milk prices are high because one loses the gross margin of the cow; and vice versa). So the mitigation costs can vary significantly, depending on actual and future market prices (which are denoted as exogenous variables by the model definition of a supply side model (SSM) without market feedbacks). These price sensitivities of mitigation options can be identified as price-related barriers of implementation and application of the mitigation strategies because they may be coupled with a high price risk. In the light of a planning horizon over several periods, the price risks of different options can have significant differences when comparing permanent options (not allowing for reaction when prices change) with flexible mitigation options (allowing for a flexible adjustment or reaction to price changes which impact the costs of the abatement strategy).

In contrast to that, other options like different stable types or manure application and storage techniques are not as sensitive to market prices. Prices that influence the mitigation costs of these options are only the prices for investment in the year of installation. The amortizations of these investments per year quantify the associated costs that are combined with the abatement caused by the mitigation options (plus possible interest payments for investment credits). Regarding this, a slight limitation has to be considered, because investment-based techniques can also lead to indirect costs induced by differences in N loss rates by different storage or application types.

Hence, different mitigation options have different levels of sensitivity to external changes. So they have different uncertainties. This has to be implemented in a proper decision framework for the choice of farm-level abatement strategies, which is also mentioned by SCHILS et al. (2005: p.174), who state that “the uncertainty itself should be used as one of the selection criteria for mitigation options”.

3.5 Implementation of abatement options into the model approach

As subsequent chapters consist of published articles, which are more or less restricted concerning the explanation of the methodological implementation of the mitigation options into the model approach of DAIRYDYN, a short illustration is given here. First of all, not all of the above-mentioned abatement options have been captured by the model to date. The stable system (slatted floor) is preset and cannot be changed by the user[20]. Carbon sequestration or release by change of tillage techniques was not part of this investigation

[20] Change of stable systems to solid floor or straw based systems is up to now not captured by the model.

either.[21] The addition of fats and oils is only recognized regarding their impact on the digestibility of the ration, which impacts the methane production of digestive processes.

The rest of the above stated options that are realizable in Germany or Europe are implemented into the analysis. A full list is given by the tables in chapter 4.5 as well as in chapter 7.3.3.

The simulation model consists of a detailed fully dynamic mixed integer linear programming model. The linear programming leads to optimization of the farm program under specific technical and economic restrictions following an economic objective function. The model, as a bottom-up approach, illustrates single defined dairy farms. GHG abatement costs are derived by stepwise decreasing the GHG constraint of the farm. Changes in profits between simulation-steps are then related to the mitigated emissions to quantify the additional abatement costs per GHG-unit.[22]

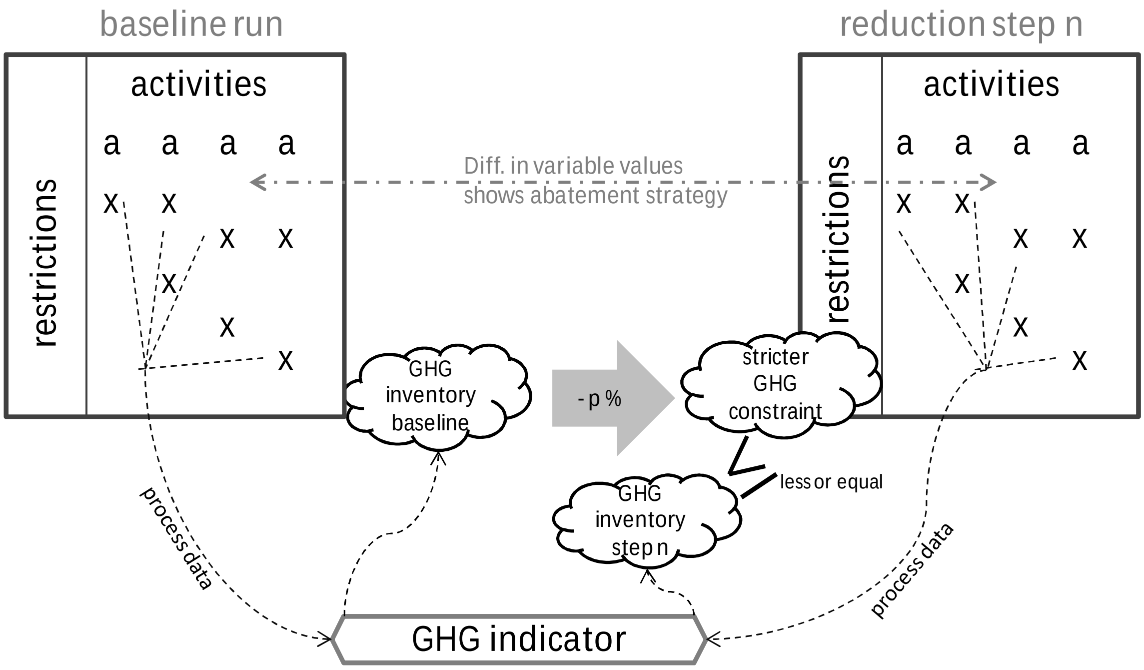
The approach does not provide a list of single isolated mitigation options with adherent abatement potentials and costs for all farms. Abatement strategies (which can consist of the interplay between more than one abatement option) are implemented indirectly by the GHG accounting indicators that are explained in the following chapter 4. As illustrated in detail in chapter 4, the different indicators couple emission factors (mostly IPCC (2006) based to guarantee a consistent accounting methodology, see chapter 4) to different production variables. Thereby, one unit of the indicator-relevant variable is loaded with a specific emission amount which implicitly shows the potential mitigation effect of lowering the level of this variable. However, as changes of one variable may also have impacts on the level as well as the emission quantities from other process variables, interaction effects may also occur (considered by the simulation algorithms of the model). Furthermore, different indicators implement different production variables into the GHG calculation; hence, only the respective indicator relevant variables are to be adjusted throughout the simulation steps under stricter GHG ceilings. It is therefore impossible to offer a unique list of emission abatement potentials and costs for single strategies as they strongly depend on specific farm characteristics that can be defined by the model DAIRYDYN and that also change throughout the simulation process. Costs as well as mitigation effects of single measures in our model approach hence depend on the system status of the farm under investigation.

[21] Different tillage techniques have been added to the model in the meantime.

[22] More explanation is given in chapter 5.

The following figure illustrates the simulation procedure to clarify how the above mentioned mitigation options are implicitly included into the linear programming model. The solver optimizes the objective value the under recognition of specific constraints. Therefore, the process variables *(x)* of each production activity *(a)* are adjusted so that a profit maximal farm plan is reached. In the baseline run, there is no GHG ceiling implemented and the solution shows the economically optimal production plan. The GHG inventory of the baseline run is then stepwise reduced by *p*% throughout the simulated reduction steps to define a binding GHG constraint. The optimization algorithm now changes variables of the production process to obtain a profit maximal production plan taking into account that the emission inventory has to be less than or equal to the implemented GHG constraint.

Figure 5: **Simulation process and implementation of GHG abatement measures** (*p*: percentage reduction of baseline emissions, *a*: production activities, *n*: reduction step, *x*: single variable of a specific production activity)



Source: own illustration.

When comparing the production variable values *(x)* (like activity levels, N use, investment decisions, etc.) of the reduction scenario with the baseline run it is indicated in which production processes changes have been made in order to yield a cost minimal GHG reduction in the required amount. These adjustments denote the abatement strategy which may consist of an interplay of single measures where isolated effects of one measure cannot be quantified exactly (because of interactions).

The way in which variables are changed in level or activities are added depends on the economic effect the specific adjustments have as well as on the mitigation potential that the specific indicators estimate for these changes in the production plan. This depends on the GHG accounting equations (chapter 4) and the implemented emission factors. Thus, mitigation potentials and costs of single strategies may strictly vary depending on the defined farm plan and the GHG indicator chosen (depending on the indicator, effects of single strategies are not accounted by default effects but individually adjusted to the single farm characteristics).

By the above-stated highly detailed modeling procedure it is possible that each farm can flexibly adjust its production processes to stricter GHG ceilings in a cost minimal way. Further on, depending on the GHG indicator, mitigation and interaction effects of abatement measures are individually adjusted to each single defined farm under investigation. More insight regarding the specific emission equations and factors is given by the indicator definition in the following chapter 4, as well as by illustrative emission factors used in the publication shown by chapter 5. The indicators illustrate which process variables are calculation relevant and thereby highlight which changes in the production process (and hence which abatement measures) are accounted for by the specific indicators.

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Chapter 4: **Construction of different GHG accounting schemes for the approximation of dairy farm emissions**[23]

Abstract

Whether for political aims or environmental aspects, quantification of greenhouse gas emissions stemming from agricultural production processes is demanded. But real measurement of greenhouse gas emissions from agricultural production is not practicable because of the diffuse sources of CO_2, N_2O and CH_4 at the farm level. This circumstance especially holds for dairy production systems, with their wide areas of cultivated soils, pasture and to a large extent open, fresh air stable systems. Hence, calculation schemes have to be constructed, enabling us to quantify an emission inventory knowing only limited attributes at the farm or sectoral level. This chapter therefore describes the details of five different emission calculation schemes, named emission indicators. They differ in variables used for emission calculation, and vary from an aggregated default formulation to a highly detailed and disaggregated construction. Basically, they are derived from IPCC methodology but with several enhancements and improvements.

Keywords: *greenhouse gases, GHG calculation, GHG indicators.*

[23] This part is based on a technical paper which was developed during the work on the DFG funded project with reference number HO 3780/2-1. It originally was part of the published article shown in chapter 7 and hence may show similarities with chapter 7. The technical paper is available on the project related web-page of the Institute of Food and Resource Economics of the University of Bonn as LENGERS, B. (2012): Construction of different GHG accounting schemes for approximation of dairy farm emissions. http://www.ilr.uni-bonn.de/agpo/rsrch/dfg-ghgabat/dfgabat_e.htm

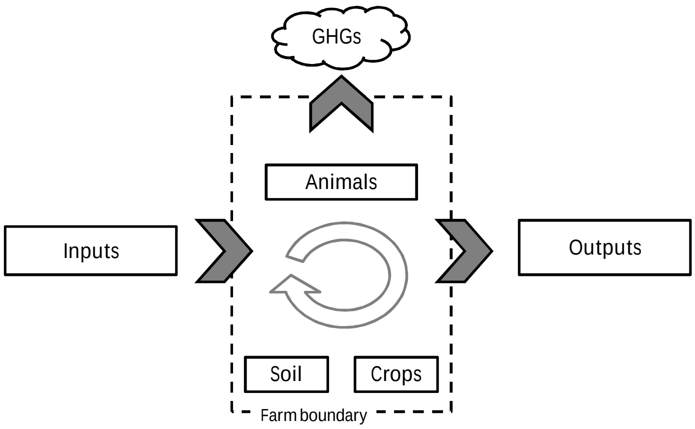
4.1 Introduction

This chapter comprises a brief explanation of different emission accounting schemes (indicators) designed for dairy farms, adjusted to the German context. Up until now, the derived calculation schemes are only applicable on dairy farms with free stalls with slatted floor. In additional to emission accounting from husbandry, associated emissions from managed manure, cultivated acreage and pasture are implemented in the estimation procedures. The content of this paper is also subject to a broader context, dealing with the derivation of abatement and marginal abatement cost curves for emissions from dairy farms. A model approach, named DAIRYDYN (LENGERS and BRITZ, 2012) has been built to derive monetary losses due to emission ceilings. To control these ceilings, emissions have to be quantified by calculation procedures. The quantification schemes implemented in the model DAIRYDYN are explained in the following sections.

4.2 The dairy farm as the system for analysis

In dairy production systems, manifold sources of CH_4, N_2O and CO_2 exist. Whereas methane mainly stems from digestive processes as well as anaerobic processes in manure storages, nitrous oxide as well as carbon dioxide mainly originate from processes in soils or from manure management as well as application of nitrogenous fertilizers. GHG accounting schemes for dairy farms have to reflect these emitting processes, and take their interactions properly into account (MACLEOD et al., 2010: p. 200). HALBERG et al. (2005: p.43) stated that "[...] the definition of system boundaries is very important for indicator selection and for interpretation of results."

Figure 1: **System boundaries for GHG emissions in a whole farm approach**



Source: own illustration following SCHILS et al. (2007: p.241).

We use the farm gate as the system boundary so that only emissions directly linked to processes on the farm are recognized (Figure 1). Emissions linked to off-farm processes such as production of purchased inputs as considered in lifecycle assessments (JOSHI, 1999) are not credited to dairy production. Our system definition fits the accounting system of the Kyoto protocol and related costs-by-cause based policies.

4.3 Necessity of proper emission indicators

Due to the "non-point source" character of agricultural GHG emissions (OSTERBURG, 2004: p.209), actual total farm emissions are impossible to measure physically. That holds especially for ruminant farms, which typically combine various cropping and grazing activities with housing of animals in stables. Measurements in ruminant stables are not only quite expensive but also hindered by air exchange via various channels (stable doors, windows, vents, fresh-air systems, etc.) and not, as in closed buildings, only via a "bottle-neck" (SCHEELE et al., 1993: p.302) such as an exhaust vent installation. Given the manifold types of stables used, it is also unclear to what extent existing measurements are representative. Further on, depending on the number of grazing hours, differing shares of the emissions from the herd or excreta occur outdoors. Accordingly, widespread direct measurement of GHG emissions in dairy farming is not practicable, so indicators are needed in order to include dairy farms into emission policy regimes. These indicators must rely on data which are accessible on the farm.

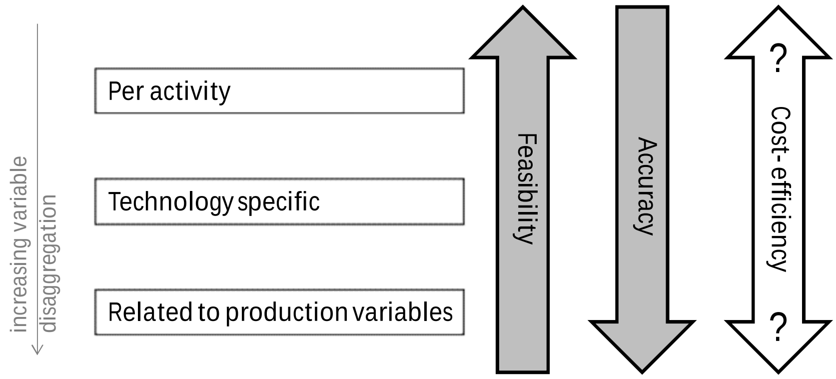
This necessary compromise between accuracy and practicability is also found in the indicator definition given by SAISANA and TARANTOLA (2002: p.5): "Indicators are pieces of information that summarize the characteristics of a system or highlight what is happening in a system. They are often a compromise between scientific accuracy and the information available at a reasonable cost." When referring in the following to a GHG indicator, we mean an accounting system that provides a GHG emission estimate from a dairy farm over a period of one or several years.

4.4 Requirements of promising indicators

The most important criteria for appropriate indicators discussed in the following section are based on findings from BACH et al. (2008: p.10), EUC (2001: p.10), KRISTENSEN et al. (2009: pp.15-16), OECD (1999: p.19) and OSTERBURG (2004: pp.210-211). They can be summarized as three criteria: *feasibility, accuracy* and *cost efficiency.*

Feasibility refers to the use of data that can actually be monitored and controlled at the farm level. As visualized in the following figure, feasibility decreases with increased requirements regarding input data.

Figure 2: **Trade-off concerning the complexity of indicators**



Source: own illustration.

Accuracy is linked to precision in emission factors and the quality of the input data. If emission factors of one variable, for example a cow, vary with decisions made on other variables like milk yield and fodder, a more detailed model that will derive the emission factors of the cows from their determining activities is more accurate than an indicator that always presumes a default emission factor per cow. SCHRÖDER et al. (2004: p.20) underline the importance of indicator consistency (e.g. by avoiding double-counting) and accounting for all relevant GHG emissions. Consistency is highly relevant for GHG emissions from dairy farming, where different gases from highly interlinked processes and sources (animals, manure storages, soil management, fertilizer practice) need to be assessed.

Cost efficiency of indicators refers to two different dimensions, the *farm-level* and the *societal* perspective. A farm faced with an emission policy instrument based on an indicator faces two types of costs: (1) monitoring costs to record and report its emissions, and (2) typically more important, costs linked to emission mitigation. Both depend on the indicator chosen. Simple indicators drawing on aggregate farm attributes such as herd size offer rather limited abatement strategies, often a single one which could provoke high abatement costs (PAUSTIAN et al., 1997: p.230), a point also raised by SMITH et al. (2007: p.22) and SCHRÖDER et al. (2004: p.20). Considering additional decision variables thus could help trigger effective and cost efficient abatement options while hopefully also improving accuracy in measuring emissions.

The costs from a societal perspective encompass, first, welfare changes in the narrow economic sense provoked by changes in farm management, i.e. profit losses to the farms, but probably also costs to consumers facing higher prices, or profit changes in up- and downstream industries or changes in tax revenues. Second, society faces costs to implement the legislation, to control the individual agent's efforts. Third, society benefits from the reduced GHGs emitted, the reason for implementing the policy. It is important to note here that an indicator not accurately reflecting changes in emissions will result in differences between private and societal abatement costs (even if measurement, administrative and control costs are excluded). This will lead to differences between cost efficiency on the farm and actual cost efficiency at a societal level, because of differences between the indicator dependent mitigation effort and the actual quantity of abatement.

Agriculture is characterized by an atomistic and heterogeneous farm structure. Indicators must hence be applicable to different types of dairy farms to guaranty cost efficient measurement and abatement options, and should, given the dynamics in farm structural development and technical progress, reflect changes in farm attributes properly.

Figure 2 illustrates the trade-off between calculation accuracy and data feasibility, which is important for the choice of a politically relevant indicator scheme (WALZ et al., 1995). The highest level of aggregation is given when emissions are calculated via IPPC Tier 1 default values, which are linked to crop acreages and average annual herd sizes. The accuracy can be improved by disaggregation: adding further attributes such as milk yield or further processes such as fertilizer application, or by disaggregating processes, e.g. in time. But the higher the complexity and disaggregation level of indicator schemes, the less available are relevant data, which restricts the indicators' feasibility.

It is obvious that an indicator needs to be based on available - financially, technically and institutionally - and reliable data (HALBERG et al., 2005), most probably preventing the best possible indicator from the viewpoint of accuracy from being chosen. It is far less clear what level of detail in a GHG indicator should be chosen from a cost efficiency perspective. Driving up the level of detail in indicator calculations increases costs for monitoring and control, but only leads to abatement cost savings if it triggers more cost effective abatement strategies.

4.5 GHG mitigation options on dairy farms

As noted before, the indicator should sensitively account for low-cost reduction activities (OENEMA et al., 2004: p.174; OSTERBURG, 2004: pp.210; CROSSON et al., 2011: p.41). The efficiency of different indicators can thus be determined by checking if promising abatement options such as shown in table 1 are properly taken into account. LENGERS (2012) provides details regarding these abatement possibilities and discusses aspects their choices are based on. Only mitigating options applicable to German dairy farms with clearly identified effects on GHGs are listed and analyzed in the following[24].

Table 1: **Applicable options to reduce GHGs from dairy production systems.**

measure	purpose
reduce CH4 emissions	
(i) variable options (flexible adaption possible)	
• improving feeding of animal	• increase animal productivity, improve digestibility, decrease CH4 from enteric fermentation and manure
• feeding additives	• decrease CH4 from manure and enteric fermentation
• pasture management	• possibility of improving digestibility of feed, lowering CH4 from enteric fermentation, lowering manure amounts in manure storages
• reduction of livestock number	• decrease CH4 from manure and enteric fermentation
• manure storage time	• prevent anaerobic conditions in manure by regular emptying of the storage
(ii) permanent options (investment based)	
• type of manure storage/coverage	• decrease CH4 from stored manure by fluxes
• stable type	• changing from slatted floor to straw based systems can lower CH4 emissions due to less anaerobic conditions of manure storage; also differences in tied stall, free stall and deep litter
reduce N2O emissions	
(i) variable options	
• change of crop rotation	• use of more N efficient crops
• reduction of livestock number	• decrease N amounts in manure
• animal nutrition	• increase animal productivity and decrease N in manure, N-reduced feeding
• restrict grazing	• decrease urine/dung excretion in the field
• adjusting N application to crop demand	• increase N efficiency of applied N fertilizers
• accounting for mineralization of organic N	• decrease required fertilizer N
• soil cultivation	• optimize growth and N uptake of crops, increase aeration and decrease denitrification
• reduction of urine N content	• decrease N2O production
(ii) permanent options	
• stable type	• change from straw to slurry based systems lowers N2O emissions
• application technique with low NH3and N2O losses	• higher N use efficiency of manure N
• storage of manure with low NH3 and N2O losses	• higher N use efficiency of manure N
• anaerobic storage of manure	• decrease nitrification and denitrification

Source: illustration following FLACHOWSKY and BRADE (2007), OENEMA et al. (2001), OSTERBURG et al. (2009).[25]

[24] E.g. antibiotics as feed additives to lower emissions is broadly discussed in literature, but nevertheless banned by German and European law.

[25] Change of stable type from slatted floor to solid floor or straw based systems is up to now not captured by the model approach but relating factors may easily be added to the below illustrated indicator schemes.

Variable options (*i*) comprise management strategies that can be flexibly changed over periods (weeks, months or single years) and adjusted to changes in exogenous production conditions. Permanent options (*ii*) have a more investment based character, leading to decisions which can induce path dependencies. Hence long term investments determine future abatement options and impact GHG mitigation expenditures. An optimal emission indicator leads to minimum abatement costs by considering all mitigation options and accounting for flexible adjustment of farm processes over time (e.g. monthly manure storage time).

4.6 Development of Indicators for the model DAIRYDYN

DAIRYDYN is a fully dynamic optimization model using mixed integer linear programming for the simulation of dairy farm development over several years, optionally confronted with emission ceilings. The bio-economic model approach has an objective function maximizing the net present value of future profits and enables the user to implement different emission accounting schemes. As the model is subject to a whole farm approach, these GHG calculation procedures may even consist of an aggregation of diverse emission calculations from different sources on the farm level (animal, soil, manure, etc.). (LENGERS and BRITZ, 2012)

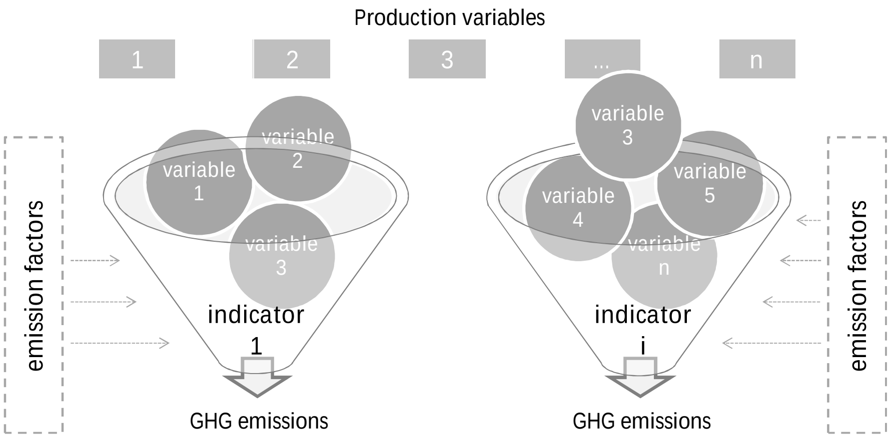
The developed emission accounting schemes are based on IPCC (2006) guidelines, which offer fundamental emission parameters and calculation schemes with accounting systems for different aggregation levels; from Tier 1, the most simple, to Tier 3 with high disaggregation and implementation of very production-specific information. These are scientifically accepted and consistent (to e.g. avoid multi-accounting bias), and have been adjusted to German circumstances and enhanced by literature findings. In the following sections the different indicator schemes are described, explaining the combination of GHG calculations from enteric fermentation, manure management, soil cultivation and fertilizer management to whole farm emission-accounting indicators (see figure 5 at the end of this chapter). I start with the simplest indicator (activity based emission calculation) and then move towards the most detailed and complex one, called the *reference indicator*. This represents the indicator with the highest degree of precision in calculating real emissions from the production portfolio of the farms. Thus it could be taken as a benchmark for valuation of the GHG accounting precision of the other indicators.

For GHG accounting, single emission factors are used, which are linked to specific production variables and quantify the proportion of gases emitted to one unit of the variable (e.g. emissions per unit of livestock) (HAUBACH, 2009: p.172). These emission ratios can also contain formulas that calculate the material conversion from input amounts (e.g. feed) to GHG release. Following HAUBACH (2009: p.172) a kind of base formula of GHG accounting can be formulated:

(1) $em_j = \sum_k ef_{jk} x_k$; with k = 1,...,n

The different indicators j link emission factors ef_{jk} to specific decision variables x_k. ef_{jk} quantifies the amount of gases emitted per unit of the variable k (e.g. emissions per unit of livestock) to derive total emissions em_j on the farm. Indicators differ in the variables used and the emission factors attached to them. For example, an indicator that only considers the number of livestock for whole farm emissions will have a different emission factor per unit of livestock than a more complex indicator that accounts for emissions from fertilizer separately. Furthermore, emission factors for the same observed attribute might also differ depending on farm characteristics (stable type, climate zone, manure management system). Figure 3 visualizes the conceptual principle of indicator schemes that use attributes of on-farm processes to quantify overall GHG emissions.

Figure 3: **Indicators calculate emissions according to different production variables**



Source: own illustration.

Obviously, emission calculation schemes differ in their data requirements: covering more attributes drives up the data demand and thus probably monitoring and control costs, while holding out the promise of improved accuracy and reduced abatement costs.

4.6.1 actBased (indicator 1)

The simplest indicator (equivalent to formula (1)) refers to the highest aggregated variable level. The single default emission factors per activity unit $ef_{actBased,k}$ in terms of arable land or livestock production are multiplied by the activity levels x_k.

(2) $em_{actBased} = \sum_k x_k * ef_{actBased,k}$

The emission parameters for CH_4 and N_2O can be derived from the IPCC (2006) Tier 1 methodology.[26] Default CH_4 emission factors for enteric fermentation and manure management per livestock unit can be taken directly from the IPCC guidelines (table 10.11, 10.14). These are defined on a regional scale for Western Europe, assuming an average stable system and manure storage techniques. N_2O emissions from specific livestock units can also be derived from the Tier 1 approach (equations 10.25 to 10.27 and relating default parameters) implementing average animal weights into the excretion function (eq. 10.30) taken from KTBL (2010) for the German context. The resulting standard emission parameters are then transformed to CO_2-equivalents according to gas specific global warming potentials and subsumed to $ef_{actBased}$ for the specific animal category. The emissions from agricultural soils are also condensed into a single default emission factor per ha of crop category. Therefore IPCC (2006) equation 11.1, accordant sub-calculations and default emission factors are used with German-specific yield levels and N requirements. For application of manure, broad spreading is assumed. Deviating from the IPCC calculations, a lower N_2O emission factor for soil background emissions[27] is used because the underlying IPCC value is based on peat soils (8 kg N_2O-N ha^{-1} $year^{-1}$). Instead, background emissions of 0.9 kg N_2O-N ha^{-1} $year^{-1}$ are taken from a study of VELTHOF and OENEMA (1997: p.351)[28]. As an improvement of the Tier 1 methodology, CH_4 background emissions are also recognized by the *actBased* indicator. These are negative and refer to the CH_4 deposition potential of agricultural soils, quantified as -1.5 kg CH_4 ha^{-1} $year^{-1}$ for cultivated acreage and -2.5 kg CH_4 ha^{-1} $year^{-1}$ for grassland (BOECKX and VAN CLEEMPUT, 2001). The improvements and enhancements of the IPCC

[26] Chapter 10 and 11 of the denoted IPCC (2006) guidelines. CO_2-C stock changes are not relevant for our overall work as no land use changes and afforestation activities are implemented in the model up to now. Also changes in tillage techniques, which would affect the carbon stocks, were not part of the GHG calculation. The C stock of cultivated land is hence assumed to stay constant over time.

[27] Named $EF_{2CG,}$Temp in the IPCC methodology (IPCC 2006, table 11.1).

[28] Multiplying kg N_2O-N by the term 44/28 results in the corresponding kg N_2O.

methodology go along with assumptions made by SCHÄFER (2006: p.156-157) for Baden-Württemberg, a state of Germany. As with livestock activities, calculated emissions from soils and fertilizer application are transformed to CO_2-equ. and summed up as an emission factor per ha of crop category, assuming average yield levels, average fertilizer use and broad spread application of manure N taken from engineering data collections like KTBL (2010).

Generally, as soil background emissions of CH_4 and N_2O are not depending on production intensity, they are taken equally for all indicator schemes (cf. figure 5).

4.6.2 prodBased (indicator 2)

The *prodBased* indicator is derived from the actBased indicator, making some adjustments concerning the variable disaggregation level. This indicator scheme also denotes differences in production output level (yield per ha or kg milk per cow) for cows and crop categories; hence, e.g. the amount of milk produced impacts the GHGs produced by one cow. So each unit of product on the farm is loaded with a product type specific emission parameter. Other sources of emissions on farms (heifers, calves, idle) which do not vary in output intensities are loaded with activity based emission parameters per ha or head taken from the actBased indicator scheme.

(3) $em_{prodBased} = \sum_k x_k * ef_{actBased,k}$; for all $k \neq crops, cows$

$$+ \sum_k \sum_p qp_k ef_{prodBased,p} \text{ ; for all } k = cows, crops$$

The overall emissions of the farm $em_{prodBased}$ use default emission factors per activity for calves, heifers and idle. Emissions produced by dairy cows, arable crop production and grassland are derived according to their specific output level qp_k of the product p which is produced by the activity k. Specific emission factors per production quantity $ef_{prodBased,p}$ of product p (e.g. emission factor per kg of milk) are multiplied by the quantity of each product per year and summed up over all product categories of activities. The product specific emission factors per unit of product are derived by taking the default emission factors per cow or per ha from the actBased scheme and dividing them by the average milk yield level per cow[29] or average yield level per ha of the specific crop

[29] e.g. a total emission amount of 3,332 kg CO_2-equ. leads to 0.56 kg CO_2-equ. per l milk for a 6,000 l cow.

or grassland. This leads to product unit[30] emission loads for each product which are taken as output level independent, disregarding effects of production intensity level on the per unit emission factor. Within the calculation of emission factors from arable land, fixed shares of fertilizer application techniques with related parameters for leaching and outgassing are assumed (comparable to Tier 1 from IPCC (2006) equation 11.1).

4.6.3 genProdBased (indicator 3)

This calculation scheme also includes the impact of the production intensity level in milk production on the emissions per kg of product. This adaptation helps to consider the development of overall emission levels as well as emissions per production unit (animal, hectare) and emission amount per unit of output, pulling in opposite directions when increasing output level per production unit. Hence, the effect of using genetic potentials in breeding activities to generate higher milk yield levels per cow can be captured by this indicator. To some extent the emission derivation is based on the former indicator schemes, using the emission factors per production output of arable production from the prodBased indicator scheme ($ef_{prodBased,p}$). Equation (4) shows the emission calculation, which comprises different calculated emission formulas for cows, crops and other production activities (heifers, calves...).

$$(4)\ em_{genProdBased} = \sum_k x_k * ef_{genProdBased,k};\ \text{for all } k \neq cows, crops$$

$$+ \sum_k \sum_p qp_{k,l} ef_{genProdBased,p,l}\ ;\ \text{for all } k = cows$$

$$+ \sum_k \sum_p qp_k ef_{ProdBased,p}\ ;\ \text{for all } k = crops$$

Following the first product of the above formula, emission amounts from heifers, calves and raised calves are also denoted by an activity specific emission factor per x_k. In contrast to the previously explained actBased indicator, these activity emission factors $ef_{genProdBased,k}$ are not IPCC default values, but are derived from IPCC (2006) functions basing on gross energy (GE) demand[31] from the cattle category (equations from IPCC (2006) chapters 10.3, 10.4 and 10.5) assuming average weights for heifers and calves taken from KTBL (2010) and GE demands derived from KIRCHGESSNER (2004). So the activity based emission factors of the *genProdBased* indicator for heifers and calves are more

[30] Product unit (e.g. kg milk, kg wheat) is not to confuse with production unit (e.g. cow, hectare).

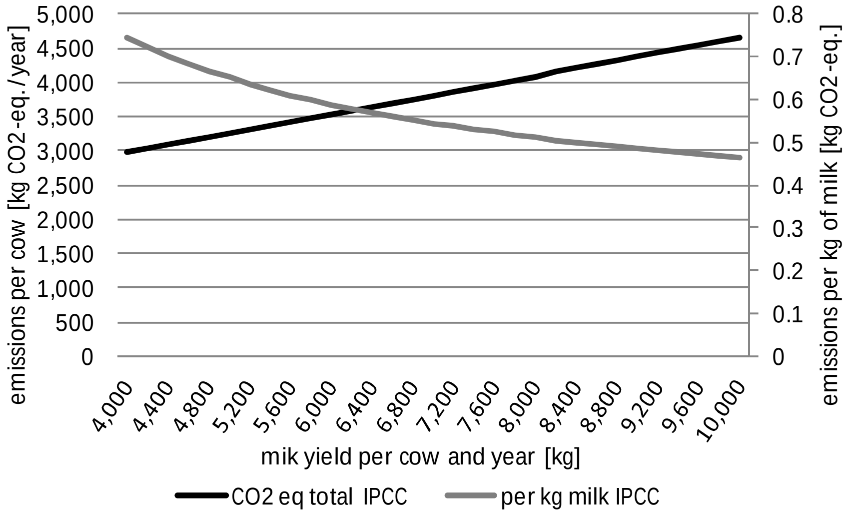
[31] For GE calculations an IPCC default value for feed energy digestibility of 60% is assumed.

adapted to real feed demand and occurring manure amounts compared to the IPCC default values.

GHG levels occurring from arable production on soils (*k=crops*) are accounted for per product unit and equivalent to the calculations of the prodBased emission factor per kg of yield ($ef_{ProdBased,p}$) from equation (3).

The greatest advantage of the genProdBased calculation scheme is the accounting of emissions from lactating cows. Emission factors per kg of milk are not constant any more, but depend on the overall milk yield level l of the specific cow. This takes the degression effect into account, which occurs from the apportionment of produced GHG emissions from maintenance and activity energy intake to different milk outputs per cow. As illustrated by Figure 4, this leads to a non-linear decrease in GHG amounts per kg of milk when the milk yield per cow increases (e.g. from 0.74 kg CO_2-equ. per kg milk from a 4000 l cow to 0.46 kg CO_2-equ. per kg of milk from a 10,000 liter cow).

Figure 4: **GHG emissions per cow and per kg of milk depending on milk yield potential**



Source: own calculation and illustration following IPCC (2006).

For GE dependent emission calculations of methane from enteric fermentation, IPCC equations 10.19 and 10.21 are used. Methane from manure management is derived from equations 10.22 to 10.24 for the output level, depending on the GE requirements of the different cow categories. For calculation of N_2O from manure management the relevant GE demand-dependent equations of IPCC (2006) subchapter 10.5 are used, assuming an average storage time of six months for manure. Following this systematic, total GHG emissions of single cows with specific genetic potential are calculated and divided by their potential milk yield per year to obtain the output level l specific emission factors $ef_{genProdBased,p,l}$ per output quantity of milk.

4.6.4 NBased (indicator 4)

The *NBased* indicator scheme describes a further disaggregated emission calculation compared to the former three indicators. Additionally, this one also accounts for differences in storage type and time and considers various manure application methods with their specific costs and impacts on emission rates of individual GHGs. In contrast to the other indicators described up to now, calculations of the NBased indicator derive GHG amounts separately for the different sources of enteric fermentation, manure management and soil management as shown by the following formula.

(5) $em_{NBased} = \sum_k \sum_l qGE_{k,l} ef_{NBased,k}$; for all k = cows, heifers, calves

$$+ \sum_m \sum_s qN_{m,s} ef_{NBased,s}$$

$$+ \sum_m \sum_k \sum_{apl} qN_{m,apl,k} ef_{NBased,apl} + \sum_k x_k ef_{NBased,back}$$; for k = crops

In the first line of the equation (5), the emissions from enteric fermentation are calculated in CO_2-equ. following equation 10.21 of IPCC (2006). For the NBased indicator, $qGE_{k,l}$, the GE demand quantity[32] by each livestock category k and each level of genetic potential l are implemented into the calculation scheme. The livestock category specific emission factor $ef_{NBased,k}$ is therefore derived by IPCC guidelines using a category specific conversion factor for methane multiplied by the global warming potential of methane (21) to yield CO_2-equivalents. Summing up over all levels of genetic potential l and livestock categories k (cows, heifers, calves) leads to the overall emissions from enteric fermentation. Variations in feed digestibility are not considered by the NBased calculation.

The CO_2-equ. resulting (from CH_4 and N_2O) from manure management before application is expressed by the second line of the above equation. Here $qN_{m,s}$, monthly (m) quantities of liquid slurry N (qN) in the different storage types s (subfloor, surface liquid storage systems without or with different coverage) are recognized to account for the manure residence time and the impacts of different storage techniques on

[32] GE demands calculated following accordant IPCC (2006) equations 10.2 to 10.16 with fix digestibility of feed.

emission quantities. The storage type s specific emission factor $ef_{NBased,s}$ is calculated on the basis of IPCC equation 10.23 to implement CH_4 emissions. Therefore an average N content of cattle slurry of 4.7 kg per m³ (KTBL, 2010) is assumed to assess the amount of liquid manure (m³) on the basis of the model given information on kg N in storage. Because IPCC formula 10.23 demands storage type specific manure quantities expressed in dry matter, an average dry matter content of 11% (KTBL, 2010) for cattle slurry is used. In order to also add N_2O emissions to the NBased emission factor for stored manure, information from IPCC equations 10.25 to 10.29 are taken to account for direct emissions and also indirect fluxes from outgassing and leaching.

Also, in cases of emissions occurring from soil cultivation (arable land, grassland) profound differences are made by the NBased indicator compared to the former ones. Background emissions from soils ($\sum_k x_k ef_{NBased,back}$) are excluded from the detailed derivation, taking standard emission factors per ha of crop or grassland quantified by -1.5 kg CH_4 ha^{-1} $year^{-1}$ for acre and -2.5 kg CH_4 ha^{-1} $year^{-1}$ for grassland (BOECKX and VAN CLEEMPUT, 2001). The other GHG emissions occurring from soil cultivation and fertilizer use (organic and synthetic) are derived from the first summand of the third line in equation (5). Monthly applied synthetic and manure N amounts to single crop categories k with different application techniques apl (broad spread, drag hose and injector, sprayer for synthetic N) are collected for emission calculation. The applied N quantities qN in month m are then multiplied by an application type specific emission factor $ef_{NBased,apl}$ and summed up over all application types, months and crop categories. This is an advantage with respect to the former indicators because manure application (time and type) can effectively diminish emissions (CHADWICK et al., 2011). Therefore, differences in emission factors for applied N to grass or arable land are also recognized. The NBased emission factor for GHG emissions from manure and synthetic fertilizer application $ef_{NBased,apl}$ is derived following IPCC (2006) equation 11.1 and relating auxiliary calculations and emission conversion factors for direct and indirect emissions[33]. To differentiate between gas release depending on manure application type (N volatized/N applied), the IPCC standard value of 0.2 (Table 11.3 from IPCC (2006)) for broad spread is changed by using assumptions from the RAINS model (ALCAMO et al., 1990) methodology to obtain lower volatilization rates for drag hose and injector application.

[33] taken from table 11.3 in IPCC (2006) guidelines.

4.6.5 refInd (indicator 5)

The digestibility (KIRCHGESSNER, 2004: p.33) of the feed consumed by each animal is also recognized by the final reference indicator (*refInd*), as digestibility is noted being the major emission reduction factor in feeding pattern (HELLEBRAND and MUNACK, 1995). Of relevance is here prevalently the energy digestibility of the ration and single supplements because IPCC enteric emissions (equation 10.21) base on gross energy demand/intake. The higher the energy digestibility, the less feed has to pass the rumen to satisfy GE demand. Furthermore, the refInd emission calculation scheme accounts for the addition of feed additives as fats and oils, as they significantly impact the energy level and digestibility of the feed ration and influence the enteric methane production potential (BENCHAAR and GREATHEAD, 2011; MACHMÜLLER and KREUZER, 1999).

(6) $em_{refInd} = \sum_k \sum_l \sum_f qFE_{k,l,f} ef_{refInd,k,f,di}$; for k = cows, heifer, calves

$+ \sum_m \sum_s qN_{m,s} ef_{NBased,s}$

$+ \sum_m \sum_k \sum_{apl} qN_{m,apl,k} ef_{NBased,apl} + \sum_k x_k ef_{NBased,back}$; for k = crops

In general, the reference indicator is constructed equivalent to the NBased indicator, dividing emissions according to their different sources (enteric fermentation, manure storage, soil cultivation and fertilizer N use). For the accounting of GHGs from manure management and arable production, the same methodologies are used as with the NBased indicator scheme (see lines 2 and 3 of equation (6)).

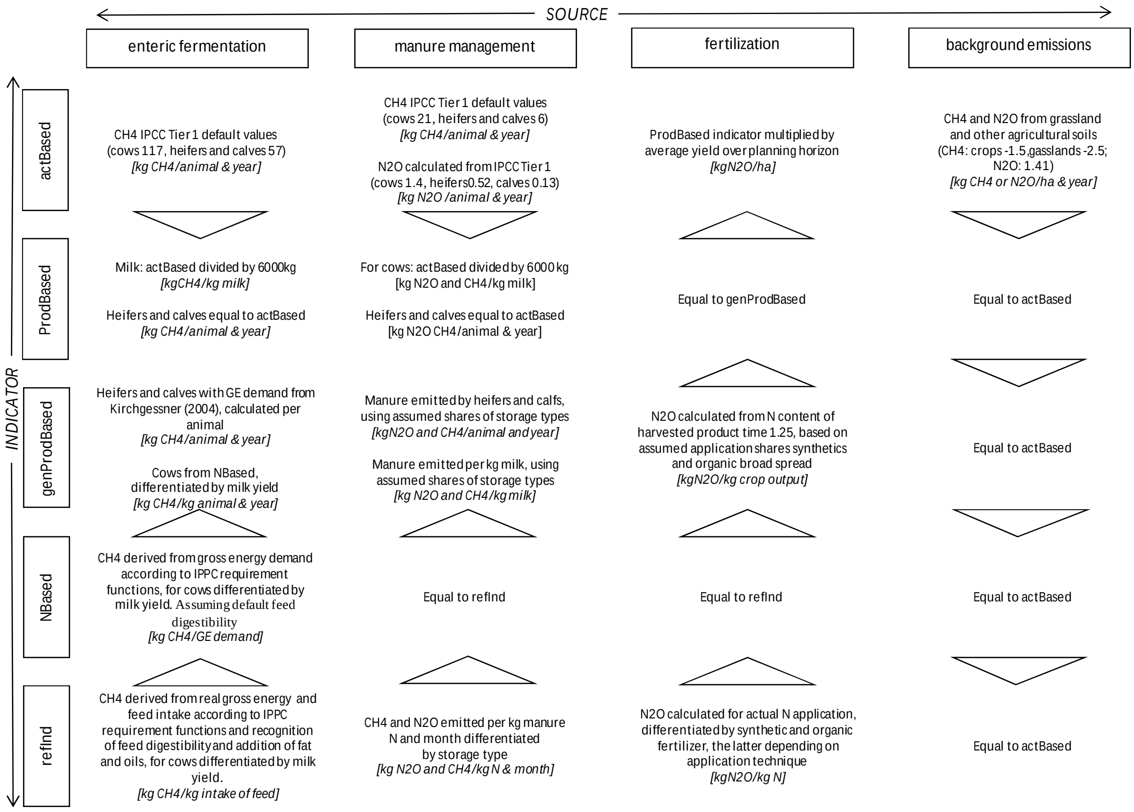
An enhancement is made by the reference indicator for the emission calculation from enteric fermentation. Line one of equation (6) visualizes that the basic principle is the same as that of the NBased indicator scheme but with minor modifications. To calculate emissions from enteric fermentation following the refInd, the derived GE demand (GE) for each animal isn't used but rather the real feed intake (FE) of several feed supplements f. GE demand, which is derived from theoretical valid demand functions, can deviate from the actual GE intake by feed in reality because of variations in feed quality and fodder availability or even if animals are reduced in intensity level so as not to fully exhaust their genetic potential (for e.g. cost efficient intensity level when prices change). Thus, $qFE_{k,l,f}$ displays the real feed intake of livestock category k with genetic potential l of feed compound f. For emission calculation the feed digestibility (di) is also recognized, leading to lower ruminant emissions for fodder rations with higher digestibility values, because less feed has to pass the animal metabolism to meet the energy demand. The emission factor $ef_{refInd,k,f,di}$ is the emission amount of methane occurring from enteric processes by implementing one kg of feed type f to the ration of animal category k. Hence, enteric emissions are also calculated following the IPCC (2006) principle in equation 10.21 but with recognition of the variability in feed digestibility and real feed intake of different compounds instead of a theoretical proper GE demand. The feed content of fats and oils, supplemented to the existing feed components, should not exceed 8% of feed dry matter (DUGMORE, 2005).

4.7 General remarks

As seen from the above stated indicator explanations, they are partly equivalent to each other in the calculation of emissions from different sources. Some of the less aggregated emission parameters are derived from the default ones used in the highly aggregated calculation schemes (e.g. the per kg milk emission factors of the prodBased indicator are derived from the default per cow emission factors from the actBased calculation scheme). Other indicator calculations constitute simplifications of more detailed aggregation schemes (the NBased calculation is a simplification of the refInd). In the case of accounting for background emissions from soils, the quantification is similar for all indicator schemes. These dependencies between the indicators' emission calculations are visualized in the following figure (Figure 5). The five indicators are plotted on the vertical axis, dividing the emission calculation from different sources on the farm along the horizontal axis.

Obviously, the different indicator schemes differ in their level of detail in implemented data for emission calculation. The more detailed the indicator scheme, the more disaggregated information from the on-farm production processes are demanded for an emission approximation, in procedural as well as time resolution. Hence, differences in the feasibility of calculation-relevant data and the resulting accuracy of emission accounting can be assumed. The feasibility and applicability of indicator schemes can be assumed to be high for very disaggregated and simple indicators. This is influenced by the availability of the required farm-level data as well as the possibility of controlling the accuracy of the collected information. Accuracy as well as induced low-cost abatement of emissions will probably increase with increasing level of detail and incorporated information used for emission approximation. The application of highly detailed emission schemes enables recognition of small differences in farm attributes and production-relevant variables. Therefore, further research must be done to quantify the emission accounting bias of the different indicator schemes. Further on, as various production variables and parameters are implemented in the calculation schemes, the indicator construction may even have decisive impacts on farm reactions to emission ceilings, which have to be explored.

Figure 5: **Overview on emission indicators**



Source: own illustration.

Overall, indicators have to be validated concerning their fulfillment of the above stated mutual requirements *feasibility*, *accuracy* and *cost efficiency* to be able to draw conclusions concerning their applicability from a political and a farm-level perspective (see chapter 7).

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Chapter 5: **The choice of emission indicators in environmental policy design: an analysis of GHG abatement in different dairy farms based on a bio-economic model approach**[34]

Abstract

The application of economic instruments to GHG emissions from dairy farms needs to rely on GHG indicators as actual emissions are impossible or extremely costly to measure. The choice of indicator impacts chosen abatement options, related costs and GHG actually emitted. A tool to quantify these relations is proposed which at its core consists of a highly detailed, dynamic mixed-integer linear programming model template able to cover a wide range of dairy farm characteristics and promising indicators. It allows deriving and comparing marginal abatement costs of GHG emissions for different farm types and indicators, informing the policy process about promising indicators, abatement strategies and related abatement and measurement costs.

Keywords: *Marginal abatement costs, emission indicators, dynamic mixed integer programming, greenhouse gas emissions.*

[34] This chapter is published as LENGERS, B. and W. BRITZ (2012): The choice of emission indicators in environmental policy design: an analysis of GHG abatement in different dairy farms based on a bio-economic model approach. *Review of Agricultural and Environmental Studies* 93(2):117-144. The original publication is available at www.necplus.eu/action/displayJournal?jid=RAE. The research was funded by a grant from the German Science Foundation (DFG) with the reference number HO 3780/2-1. Thanks also to two anonymous reviewers as well as the editor of the RAEStud journal for helpful suggestions and a straight forward reviewing process. A former version of this study was presented at the 2011 EAAE PhD-workshop in Nitra, Slovac Republic.

5.1 Introduction

Agricultural production directly accounted for 13.5% of total global greenhouse gas (GHG) emissions (CH_4, N_2O, CO_2) in 2004 (IPCC, 2007: p.36) stemming from ruminant fermentation, fertilizer use and further farm processes. With 4% on global totals, more than a quarter of agricultural emission stems from dairy production alone (FAO, 2010: p.32) which is thus an important emitter of GHGs (STEINFELD et al., 2006: pp.78; FAO, 2009: p.62). It is obvious that higher emission reduction targets, also for industrialized countries such as Germany, will require an inclusion of agriculture into GHG emission abatement efforts (e.g. BMELV, 2010: p.4), and, especially in Germany, dairy farming will be one of the key sectors.

From an economic viewpoint, promising policy instruments to steer abatement efforts are price-based such as emission taxes or tradable emission rights. Facing such instruments, firms will abate emissions as long as marginal abatement costs are lower than the emission price – which is either equal to the per unit tax or to the price of a tradable permit. Once the marginal abatement costs exceed the price, firms will either pay taxes or buy additional permits.

Accordingly, two main questions arise for an adequate policy design when targeting GHG emissions from dairy farms. Firstly, judging how costly certain reduction targets are, requires knowledge about marginal abatement cost (MAC) curves for GHG emissions of CH_4, N_2O and CO_2 for single dairy farms and for the dairy sector. And secondly, an appropriate emission indicator is needed which can be implemented at farm level to account for GHG emissions. These two aspects are strongly interrelated, as the MACs will to a large degree depend on the chosen indicator.

But why is that the case? Such as we only pay income taxes on declared income, dairy farmers will only pay emission taxes on declared emissions. The abatement strategy of a farmer and the related costs will hence depend on how emissions are defined by the specific indicator chosen – a kind of GHG tax code –, and not on the physically emitted GHGs. Options which change emissions but are not accounted for will not be integrated in abatement efforts, even if they are less costly. If measurement of GHGs would be costless, we would not need an indicator, and there would be no difference between accounted and emitted GHGs. But GHGs from dairy farms are impossible or rather costly to measure due to the "non-point source" character of agricultural production which takes place in open, human managed biological systems (OSTERBURG, 2004: p.209). Accordingly, any policy instrument targeting GHG emissions from agriculture will have to rely on GHG indicators and to face the problem to find a balance between measurement and abatement costs in relation to real reductions of GHGs.

Indications on how to construct indicators can be drawn from promising GHG abatement options discussed in literature – a good indicator should take those options into account. Changes in animal diet, manure management and control of production intensity are possible examples of such options. However, studies analyzing abatement costs so far often use rather simple indicators which are based on activity levels where GHGs of the farm are calculated by multiplying herd sizes and acreages by a fix per head or ha emission factor (BREEN, 2008: p.4; PÉREZ and HOLM-MÜLLER, 2007). These indicators are rather rough and do not account for promising abatement options, whereas fodder intake (DECARA and JAYET, 2000, 2001, 2006) or milk yield per cow are more precise and closer to the scientifically discussed abatement options. But especially fodder intake is also difficult to control.

Consequently, the questions resulting from the above stated problems are: (1) What are promising abatement options of GHGs in dairy farming? (2) What are the abatement and measurement costs for different types of dairy farms and the dairy sector as a whole under different indicators and emission targets? (3) What is an appropriate methodology to derive these costs? And (4), what drives the abatement costs under different indicators?

The objective of this paper is to present a core element of the methodology to answer these questions: a farm-specific economic simulation model which is able to cover a great variety of GHG abatement options and to derive farm specific marginal abatement cost curves for different emission indicators. Illustrative differences in MAC shapes depending on farm characteristics and indicators will be shown, using four different farms (differentiated by starting herd size and milk yield) under four GHG emission indicators. Furthermore, the paper will give first indications for the cost effectiveness of different indicators related to abatement efforts in dairy production.

The main contribution of this paper is the presentation of a highly detailed farm-specific bio-economic model which incorporates major technological and financial interactions in dairy production and allows simulating economically optimal abatement strategies under different emission indicators and emission targets. The modeling approach must capture core characteristics of dairy farming. One key characteristic are long lasting investments in stables and milking parlors which account for a larger part of production costs. The model template must hence cover a longer planning horizon. Secondly, the (bio-) dynamic character of dairy production must be taken into account. Variables like e.g. biomass, herd size and distribution of milk yield in the herd as well as existing firm endowments such as stables, machinery, equity or property rights to land or subsidies are to a larger extent state variables which are not or only to a certain extent controllable in period t, but depend on control variables of former periods (KENNEDY, 1987: p.59). In addition competitiveness, asset fixity and rapid technological change are characteristics of agricultural production (RAUSSER and HOCHMAN, 1979: p.2). Thirdly, decision variables are partly continuous (e.g. amounts of fertilizer, cropping land) and partly not (e.g. investment or labor use decisions). Therefore a dynamic mixed integer programming model approach (MIP) as proposed by NEMHAUSER and WOLSEY (1999: pp.3) and POCHET and WOLSEY (2006: p.78) is to be used to respect also integer or binary decision variables.

The following section will provide a short literature review, discussing the state of the art in the research field of deriving marginal abatement costs for GHG emissions in agriculture. From there, features of the proposed model template will be motivated. Subsequent sections focus on specific modules of the model template and their relations. After a detailed description of how abatement and marginal abatement costs are derived, farm characteristics of our illustrative simulation runs will be delineated. After discussing the resulting outputs, we will summarize and conclude, specifically regarding further research activity.

5.2 Literature review

A detailed comparison of different model approaches for the derivation of MACs for GHG emissions is found in VERMONT and DECARA (2010). They point out that so-called supply side models are best equipped to model what is normally understood as MAC curves because of their relatively detailed technological description. Many studies estimate MAC curves based on supply side models for European agriculture only considering changes in activity levels (e.g. BREEN, 2008: p.4; DECARA et al., 2005: pp.559). Besides herd size changes or changes in cropping area, DURANDEAU et al. (2010: p.58) also took adjustments in fertilizer use into account when evaluating abatement costs for reducing N_2O-emissions from soil in an application to a French region. In an EU-wide application, PÉREZ and BRITZ (2010) considered changes in herd size, yields, cropping areas and fertilizer practice. A model approach that already implements more detailed emission calculations, based also on ruminant fermentation, feed intake and fodder composition is presented by DECARA and JAYET (2000). The authors developed a linear programming (LP-) approach for French agriculture to evaluate GHG abatement costs which has been subsequently improved (2001, 2006). The mentioned studies model either a regional aggregate of all farms or aggregate of farm types for rather large regions, carrying the risk of aggregation bias (PÉREZ et al., 2003: p.7) and do not allow analyzing in detail differences evolving from farm characteristics.

Equally, the approaches are comparative static so that dynamic aspects e.g. relating to herd management and investments are not taken into account, carrying the risk to overestimate MACs. In Europe, HEDIGER (2006) incorporates abatement options in a recursive dynamic modeling exercise to consider investments and further time dependent aspects in an application to whole Swiss agriculture. The results underline that investment-based abatement options should be considered, requiring a dynamic perspective as offered by dynamic programming. An example for a dynamic approach relating to herd management is presented by HUIRNE et al. (1993) for replacement decisions of sows, but the basic structure can easily be transferred to dairy cows.

Existing studies calculate emission abatement costs given a specific GHG indicator, not investigating differences between GHG emissions, abatement strategies and costs under different indicators. Only DURANDEAU et al. (2010: p.71) highlight that the choice of the emission indicator is a key question in the design of emission policy schemes as it will have a strong influence on abatement, implementation and monitoring costs. As underlined in the introduction, a cost-effective abatement is strongly dependent on the design of an emission indicator, but studies which discuss and compare varying indicator systems for GHG abatement in agriculture do not exist.

WEISKE and MICHEL (2007) show modeling results of different abatement strategies in dairy production based on an economic engineering model. They evaluate the abatement potential and related costs of different feed mixes and conclude that promising GHG-reducing feeding strategies depend on farm characteristics. Accordingly, an appropriate modeling approach should allow for endogenous and variable adjustments of the feed mix while properly reflecting the impact of feed mix changes on emitted GHGs.

In order to improve on existing studies, promising abatement options for GHG for dairy farms need to be collected and integrated in the model template. Abatement strategies that are mentioned in literature (e.g. by BATES, 2001: pp.11; FLACHOWSKY and BRADE, 2007: pp.424; GUAN et al., 2006; JENTSCH et al., 2007; JOHNSON et al., 2007; KAMRA et al., 2006; KTBL, 2002: pp.203; MCGINN et al., 2004; OSTERBURG et al., 2009: pp.49; UNFCCC, 2008: pp.18 and WEISKE, 2006) range from variable feed adjustments to investment decisions for manure coverage. To evaluate the different options, studies like BOADI et al. (2004: p.330) give a qualitative benchmark of the practical availability and feasibility of the different strategies, here for the abatement of methane. Emission parameters and emission functions linked to production activities will be based on literature, e.g. IPCC (2006) and DÄMMGEN (2009). Several studies do not only list abatement options, but also quantify reduction potentials and related costs, e.g. WEISKE (2006). In the following the methodology and construction of the model is described. After the explanation of interactions between the different dairy farm production modules, the derivation process of MACs is described. An analysis of illustrative model results will complete this part and highlight areas of further research and model expansion.

5.3 Methodology

Our single farm model template, named “DAIRYDYN”, is based on mixed-integer, fully dynamic linear programming. A programming approach allows describing in great detail the technological relations between different decision variables as discussed below. Integer decision variables are necessary to account for the non-continuous character of labor use and investment decisions. Furthermore a fully dynamic approach is deemed important to account for both the forward looking character in developing farm business plans incorporating long-lasting investments and the strong inter-annual dependencies in dairy herd management. It allows depicting factors impacting the development of dairy farms independently from GHG related policy instruments (e.g. breeding to higher milk yields per cow) which might also change GHG emissions. A fully dynamic approach allows comparing baseline developments (without emission ceilings) against those under emission reductions to identify GHG abatement activities that are implemented additionally. Otherwise, GHG abatement activities additional to ongoing processes may be obstructed (SMITH et al., 2007: p.8).

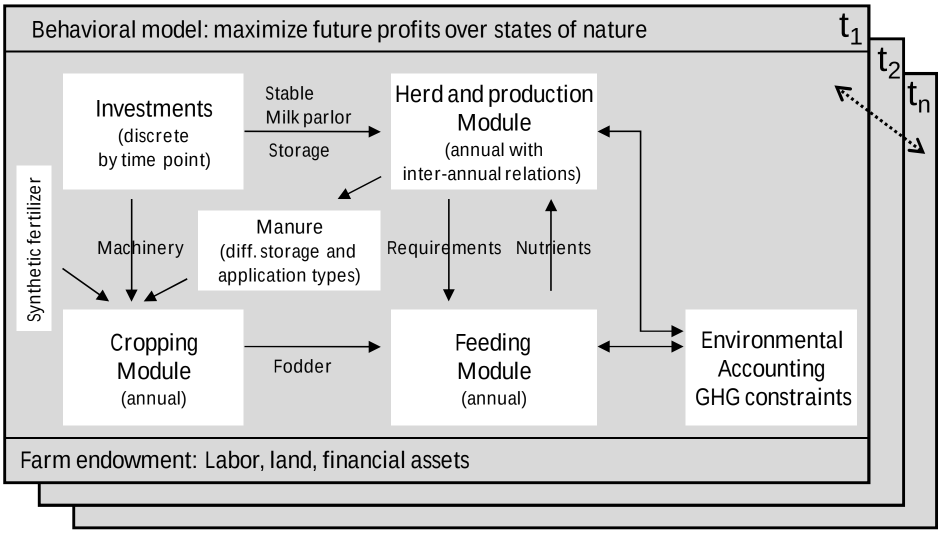
5.4 The Model

5.4.1 Overview

We assume a fully informed, rational, risk neutral decision maker maximizing net present value of expected profits under different states-of-nature with given probabilities. The states of nature currently relate to different key prices (milk, beef, concentrates) faced by the farmers. The famers draw revenues from subsidies (single farm payment), selling farm products (cash crops, milk, calves, slaughtered cows) and selling or renting out assets (land, interest on equity, off-farm labor), while facing expenditures from buying inputs (fertilizer, concentrates, labor ...) or investment goods (land, machinery, stables), from paying back credits and interest on them, as well as from given household expenditures. A positive cash balance has to be maintained, if necessary by external financing. The accumulated cash balance minus open loans at the end of the planning horizon is the objective value.

The model template consists of different modules describing sub-systems of a dairy farm level. Figure 1 visualizes these modules and specific interactions between them over several time periods (t_1-t_n) depending on the relevant planning horizon.

Figure 1: **Overview on model template**



Source: own illustration.

The *herd and milk production module* covers decisions concerning replacement of cows, growth or reduction of herd size as well as changes in milk yield of the herd. Female herds (dairy cows, heifers, female calves) are differentiated in strata by their maximum milk yield. A dairy cow with a given milk yield potential gives birth to calves with a different milk yield potential from which the farm can select, depending on transition probabilities between generations. The model thus describes endogenously the development of the milk yield potential in the herd. A sharper selection reduces possible herd expansions (at least in the current version where females cannot be bought). At the same time, cows with a higher milk yield are characterized by a lower number of lactations and higher labor needs, and as discussed in the next paragraph, by different feeding requirements. Decisions in herd module are closely interlinked with the feeding module.

The *feeding module* consists firstly of requirement functions (energy, protein, max and min dry matter and fibre etc.) for each herd. For dairy cows, these requirements are defined for different lactation periods and depend on the average daily milk yield in these periods. Secondly, it comprises endogenous variables which distribute feeding stuff to livestock categories which need to cover livestock nutrient requirements. These variables are differentiated by herd, year, lactation period and intra-yearly planning period.

The *cropping module* describes land use, distinguishing between arable and grassland activities. The latter are differentiated by intensity (number of cuts and grass yield) and management type (grazing or cutting). Grassland activities deliver certain amounts of grass in different intra-yearly planning periods. Cropping activities demand machinery - link to the investment module - and labor, and are characterized by costs and, if applicable, market revenues. Furthermore crop nutrient requirements and balances are introduced to model endogenously the application of mineral fertilizer and manure.

The *investment module* covers endogenous decisions about investments in new stable places or milking parlor, liquid manure reservoirs and machinery. Additionally, the template captures labor by intra-annual planning periods, which allows farm family members to work off- or on- farm and to hire external labor.

The *fertilizing and manure handling module* depicts synthetic fertilizer use and manure handling, in the latter case capturing different storage types (subfloor or in surface reservoirs), the possibility to cover surface reservoirs with straw or foil and different application techniques. These details are introduced to account for NO_x and further N-losses dependent on stable, storage and application type.

Wherever necessary and applicable, decision variables are linked to emission parameters for CH_4, N_2O and CO_2. That means that selected variables of the model carry emission factors according to the applied emission crediting system (GHG indicator) to calculate endogenously an overall GHG amount from the production program of the farm.

Attention is paid that the different modules cover relevant abatement options for GHGs discussed in literature (e.g. increasing milk yield per cow, investments in certain stable types, manure storage coverage, use of feed additives, changes in feed mix and variation of herd size) with their specific mitigation parameters, their interactions, the associated costs and further attributes for e.g. labor need or content of feed stuff. Simulations with the template then also take indirect impacts of these options on the farm program (e.g. changes in the feed mix impacting crop shares, crop management and manure management) and thus profits into account.

To build up farm models with a highly disaggregated production process of dairy farming, information are taken from detailed farm management handbooks such as KTBL (2008, 2010) which also cover investment costs for machinery, building and other farm equipment. Abatement simulations are based on GHG emission restrictions which determine an upper limit for GHG emissions of the whole farm. These are defined based on decision variables and attached GHG emission factors, the latter depending on the specific emission indicator chosen. New or stronger restrictions might require adjustments in farm program. The resulting changes in farm profits are then used to derive abatement and marginal abatement costs, specific for the farm, the indicator and the GHG reduction level. This process will be described more detailed in section 5.5 in this chapter.

5.4.2 Detailed presentation of specific modules

Herd and production

The herd module captures different decision possibilities to control herd size and milk yield during the planning horizon. It has an annual resolution and differentiates between dairy cows, heifers and female calves for replacement and female and male calves sold. Dairy cows, female calves and heifers for replacement are further differentiated by their potential milk yield. Consequently, in any one year, the herds simulated for a farm will typically consist of different groups of dairy cows, female calves and heifers for replacement differentiated by their potential milk yield. Starting with the initial herd with a specific genetic production potential, cows give birth with a certain probability to calves with different milk yield potentials, which partly exceeds the genetic potential of the mother. The model can endogenously choose how many females of a specific potential are raised for replacement or sold. This allows hence depicting the trade-off between sharper selection and herd size increase. The calves born in a given year replace cows three years later, introducing inter-annual relations between the groups of different milk potential over time. Cows reaching their maximum number of lactations, which decrease with increasing milk yield potential, need to be slaughtered; additional slaughter is possible to reduce the herd size. In order to retain a flexible intensity management the genetic milk yield potential needs not to be fully exhausted (e.g. to manage years where fodder availability is low or feed prices are high). Furthermore, labor and feed requirements (see below) and other costs for dairy cows are differentiated by potential milk yield.

Feeding

Requirement functions are specified for the different herds according to IPCC (2006). For cows, to give an example, requirements depend on animal weight, actual fat corrected milk yield, the latter differentiated in 200 kg steps, and are specified for 5 lactation periods (30-70-100-105-60 days, where the last 60 days are the dry period) with different average daily milk yield. The functions depict energy, protein, fibre min/max and dry matter min/max, respecting the rumen capacity. In addition, max/min of certain feed are defined. These requirements enter constraints in the model template, differentiated by year, state of nature (SON) and herd – for dairy cows differentiated by milk yield -, lactation period and intra-yearly planning period, the latter to take into account available fodder from grazing. These constraints need to be covered by feeding activities which are either linked to fodder production and thus cropping activities or purchases of concentrates. The feeding blocks consequently comprise a very large number of endogenous variables. Whereas the farmer takes yearly decisions about herd size and composition only in averages over the SONs, feeding can be flexible adjusted to the SON.

Cropping

The cropping module covers different cropping activities for arable and grassland. Cash crops on arable land such as cereals or oilseeds compete with fodder production like maize silage. On grassland, silage or pasture in different management intensities are considered. The farmer can sell, buy or rent out land. The crop mix is restricted by maximum rotation share for each crop, where deemed appropriate. Cropping decisions are differentiated by crop, year, SON and, where applicable, management intensity. Yields in pasture are differentiated by planning period and, together with other types of fodder production, directly interact with the feeding module. Crops are further characterized by exogenously given labor and fertilization needs for nitrogen, other operation costs, yields and related prices, the latter can be differentiated by SON. Furthermore, the activities in the cropping module demand certain amounts of machinery available, which have to be acquired if not yet in the inventory. The above described herd and production module produces different amounts of slurry, depending on herd composition and sizes and the stable system.

Manure handling and fertilization module

The module deals with different manure storage as well as mineral and organic application techniques which might differ in NO_x emissions, providing a further link between the herd and cropping modules. Manure excretions can be either stored sub-floor or in differently sized surface manure reservoirs and the farm has to maintain certain storage capacity in relation to yearly manure output. The silos can be additionally covered by straw or foil to reduce emissions during storage. Manure can either be distributed based on spreader, a drag hose or injected. Maximum application rates and periods where manure application is forbidden are taken into account according to the German implementation of the Nitrates directive. Further on, depending on the crop, further periods might be blocked for manure application (e.g. applications after maize has reached a certain size). Besides manure, synthetic fertilizer can be used to cover plant nutrient demands.

Investments and finances

Investment decisions are implemented as binary variables with a yearly resolution[35]. Whereas feeding and cropping decisions are rather flexible and can be adjusted to changes in prices, we allow decisions upon herd size and composition as well as upon investments only in average of the SONs. Cropping activities require certain machine hours of e.g. tractors and ploughs which have to be replaced when their maximum of operation hours is reached. Different stable types (for calves, heifers, cows) in differing sizes are offered by the model to allow for building up new herd capacities or to replace old stables which have reached the end of their useful life (30 years lifetime). Stable types differ in investment costs and labor hours per stable place. As mentioned above, surface manure reservoirs are offered in different sizes and coverage techniques. The demanded machinery by the cropping activities as well as investments in buildings can be financed either from accumulated cash or credits. The latter are differentiated by pay-back time and interest rate. Accumulated cash draws interest. It is assumed that stables cannot be sold and that the demolition costs of the stables at the end of their usage equate the residual value of sellable technical equipment.

5.4.3 GHG indicators

In dairy production, manifold sources of GHG emissions exist. According to IPCC guidelines and the way the European emission trade scheme is implemented, only direct emissions of CH_4, N_2O and CO_2 from on–farm processes are accounted for in the model. The system border is hence the farm gate, so that results should not be confused with lifecycle-assessment.

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[35] It is possible to restrict investment decision to specific years to keep the number of binary variables at a manageable size.

Enteric fermentation as well as manure management are the main sources of CH_4 in dairy production systems with the majority stemming from digestive processes. Nitrous oxide emissions primarily stem from processes in agricultural soils after N application of fertilizers or during crop growth and chemical N conversion processes in soils. As N_2O production is an aerobic process and manure is mainly anaerobe, only minor amounts of nitrous oxide emissions are caused by manure storage or application. CO_2 is assimilated by crop lands and also emitted by soils if e.g. permanent grassland is ploughed. So far, CO_2 assimilation by crops is not implemented in the model[36], but following BOECKX and VANCLEEMPUT (2001) CH_4 deposition by agricultural soils is accounted for. So depending on the cultivation of land, soils can become a net source as well as a sink over a full year.

All decision variables in the model template might carry an emission factor expressed as CO_2 equivalents (single gas emissions of N_2O and CH_4 multiplied with global warming potential of 310 for N_2O and 21 for CH_4 (UBA, 2009: p.57)) and thus enter the GHG emission constraint. The emission factors are either directly taken from literature, calculated based on literature based emission functions or, in future, based on measurements at an experimental farm of Bonn University. A specific set of emission factors is termed a GHG emission indicator and thus represents a specific accounting system for GHGs from dairy farms. The minimal profit loss and related farm program under a GHG ceiling depend on the interaction between the decision variables and that ceiling via the emission factors. As depicted in the objective of this paper, different emission indicators are to be analyzed concerning their impact on the shape of MAC curves and related abatement strategies. These indicators are more or less complex and accurate. They also relate to different decision variables (number of cows, milk yield per cow, C and N in feedstock, arable activities, fertilizer intensity...) and thus determine the possibilities of farmers to react to emission ceilings. Figure 2 depicts an overview on the indicators.

The different indicators are mainly based on the IPCC (2006) guidelines[37] which comprise so-called tiers of increasing complexity to calculate GHG emission. Tier 1 provides the simplest approach to account emissions using default parameters e.g. per animal. We use Tier 1 as far as possible to define our simplest indicator termed *actBased*, where emission factors are linked to herds and crop hectares, only. The exemptions from

[36] No land use change, afforestation and change of tillage techniques implemented in the model. CO_2 calculation is thus neglected up to now.

[37] Equations and parameters of sections 10 and 11.

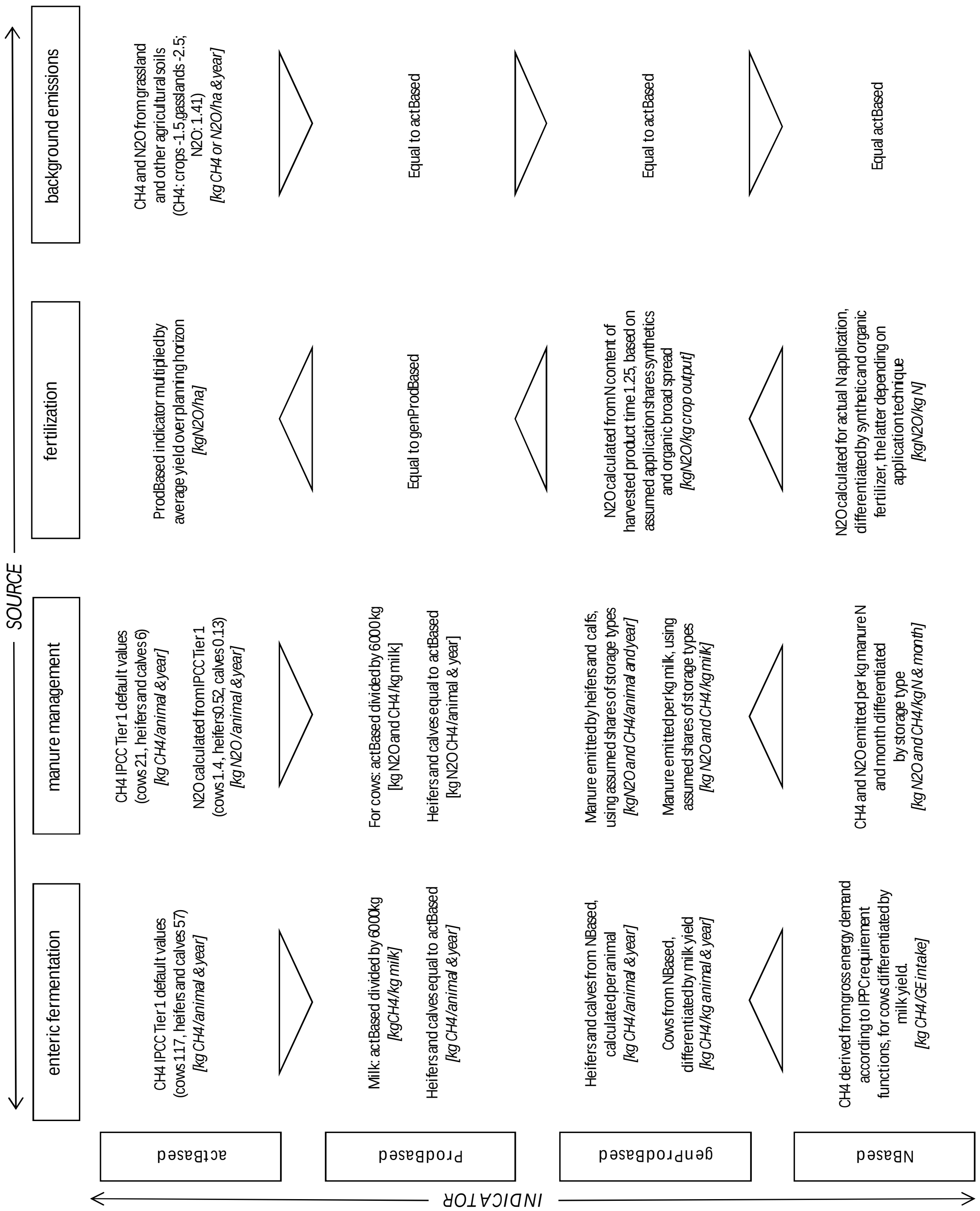
the IPCC methodology are manure management and fertilization where IPCC links emission factor to organic and synthetic fertilizer amounts. We thus assume average excretion and fertilizer application rates to derive per animal or per ha coefficients.

A somewhat more complex indicator called *prodBased* links emission factors to production quantities of milk and crop outputs, see details in table 1 below. Generally, at the assumed average yields, the two indicators yield the same overall emissions. Compared to the activity based indicator, famers have somewhat more flexibility as they might e.g. switch between different grass land management intensities to abate emissions.

The most complex and also presumably most accurate indicator is called *NBased*. Values for enteric fermentation are calculated from the requirement functions, for energy based on IPCC guidelines, which also drive the feed mix. For manure management, emissions are linked to the amount of manure N in specific storage types in each month. For fertilization, the emission factors are linked to distributed nitrogen differentiated by application technique. The indicator thus gives the farmer the chance to abate nitrogen losses by changing storage types, storage periods or the fertilization application technique, beside changes in herd sizes, herd structure or the cropping pattern.

An intermediate indicator between the prodBased and NBased one is called *genProdBased*. Its emission factors are linked mainly to output quantities but as far as possible derived from the NBased one assuming fixed application shares of synthetic and organic N. The differences, as seen from figure 2, stem from the calculation of emissions from enteric fermentation and manure management. Specifically, the indicator introduces milk yield dependent emission factors which reflect that higher milk yields reduce per liter emissions by distributing the maintenance need of the cow over a larger milk quantity, diminishing from 0.81 kg CO_2-equ. per kg milk for a 4000 liter cow to 0.40 kg CO_2-equ. per kg milk for a 10000 liter cow (see table 1). The yield level dependent output coefficients per kg of milk are hence the major advantage of the genProdBased indicator compared to the prodBased one.

Figure 2: **Indicator Schemes**



Source: own illustration.

No difference in emission calculation between the indicators is made for the background emissions coming from soils as seen from the figure 2. The chosen default values per ha are taken from DÄMMGEN (2009: p.315) and VELTHOF and OENEMA (1997: p.351). Obviously, moving from the bottom of figure 2 to the top, the aggregation level of emission relevant model variables increases which means a loss in detail concerning the decision variables addressed by the indicators.

Table 1: **Emission parameters in kg CO_2-equ. by indicator**

	calculation unit	actBased	prodBased	genProdBased	assumed av. Yield per ha or head
Cere	ha	2020.83			
Cere	prodQuant		241.48	241.48	8.3 t
Oils	ha	1585.08			
Oils	prodQuant		505.09	505.09	3.1 t
Rest	ha	588.47			
Rest	prodQuant		140.64	140.64	4.1 t
MaizSil	ha	1864.70			
MaizSil	prodQuant		41.44	41.44	44.9t
idle	ha	406.93	406.93	406.93	
grasSil	prodQuant		46.62	46.62	25.4 t
gras20	ha	1188.70			
gras29	ha	1188.70			
gras34	ha	1188.70			
past33	ha	2660.32			
grasPasture	prodQuant		80.62	80.62	32.9 t
milk	prodQuant		0.56		6000 kg
cows4000	prodQuant			0.81	
cows5000	prodQuant			0.67	
cows6000	prodQuant			0.58	
cows7000	prodQuant			0.52	
cows8000	prodQuant			0.47	
cows9000	prodQuant			0.43	
cows10000	prodQuant			0.40	
mCalvs	head	52.29	52.29	27.12	
fCalvsSold	head	52.29	52.29	27.12	
fCalvsRais	head	1363.30	1363.30	707.00	
heifers	head	1484.20	1484.20	1358.50	
cows	head	3332.00			

Source: own calculation and illustration following IPCC (2006) and DÄMMGEN (2009).

The parameters for the three simpler indicators are shown in the table above. Computations for the NBased indicator are also taken from the IPCC (2006) guidelines, focusing on equations from subsections 10 and 11. For the direct emissions from managed soils equation 11.1 is taken with the corresponding auxiliary calculations and default emission parameters. Equations 11.9 and 11.10 are used to derive indirect emissions from soils, only the default values for background soil emissions (N_2O) are taken from VELTHOF and OENEMA (1997) due to a correction in emission level[38]. Emission calculations from enteric fermentation and manure management are also based on IPCC stemming equations (subsection 10), using where possible also Tier 2 equations.

[38] IPCC default value is 10 times higher because the underlying study bases on peat soils.

5.5 Derivation of marginal abatement costs for single firm

Under a given indicator, a stepwise reduction of the emission constraint will potentially lead to a stepwise reduction in farm profits. Relating the change in emissions to the changes in profits allows calculating the total and marginal abatement cost.

In the following, em_{0j} are the emissions measured with indicator j under the profit maximal farm program without any emission target, where the zero characterizes the reduction level. The reader should note that different indicators are attaching different GHG emissions to the very same farm program.

To derive marginal abatement cost curves, an emission ceiling will be introduced and stepwise lowered. n reduction steps, each with the same reduction relative to the base em_{0j}, will be taken, leading to objective values from π_{0j} to π_{nj} (where π_{ij} is the value of the objective function in simulation step i, using indicator j; with i from 0 to n). Let rec_i denote the emission ceiling in step i relative to baseline emissions. The maximal profit under the derived absolute ceiling $rec_i em_{0j}$ is restricted according to:

(1) $\sum_k ef_{jk} x_k \leq rec_i em_{0j}$

Where x_k are the decision variables and ef_{jk} the emission factors attached to them under indicator j, i.e. the CO_2 equivalent emission accounted per unit of variable k.

The difference in profits between π_{0j} – the profit without a GHG restriction – and π_{ij} measures the profit foregone due to ceiling $rec_i em_{0j}$ and defines hence the total *abatement costs* (AC) for the reduction level of step i and indicator j:

(2) $AC_{ij} = \pi_{0j} - \pi_{ij}$

A stepwise reduction of the emission constraint leads to a sequence of changes in farm program and related profit losses. Relating these differences in profits to the difference in emissions defines the simulated *marginal abatement costs* (MAC):

(2.1) $MAC_{ij} = \frac{\pi_{i-1,j} - \pi_{i,j}}{em_{i-1,j} - em_{i,j}}$

When comparing different emission indicators we face the problem that the MACs of each indicator relate to its specific GHG accounting rules. Accordingly, the MACs of different indicators cannot be compared directly.

From a policy perspective, we would like to assess costs and benefits of choosing a certain indicator and ceiling based on the GHGs physically released from the farm, and not the GHG accounted by a specific indicator. Indicators might over- or underestimate physical GHG emissions and thus under- or overestimate the "true" MACs.

In an ideal world, we would be able to derive the "real" GHG emissions from the farm program. As this is impossible, a so-called *reference indicator* will be constructed. It will use the best available scientific knowledge to derive from the farm program, i.e. based on all available decision variables, a total GHG emission estimate from the farm. The underlying calculation could be highly non-linear and complex and need not necessarily be integrated in the model template itself. Equally, it does not matter if it could be implemented in reality on a dairy farm given its measurement costs. It simply serves as a yard stick to normalize GHG emissions from different, simpler, but more realistic and applicable indicators. Relating profit losses under different indicators and indicator-specific GHG emission targets to the GHGs abatement under the reference indicator r at the simulated farm program allows deriving normalized marginal abatement cost curves which can be compared between indicators:

$$(2.2) \quad MAC_{i,j}^{norm} = \frac{\pi_{i-1,j} - \pi_{i,j}}{em_{i-1,r} - em_{i,r}}$$

This will show under which indicator the highest efficiency will be obtained, meaning that "real" abated emissions of the optimized production portfolios of the farms are calculated and related to the abatement costs caused by different emission indicators. Currently, we use the NBased indicator defined above as the reference indicator.

According to the stated objective of this paper, we formulate a few hypotheses and test them with illustrative model applications:

1. MACs depend on farm characteristics.
2. The model creates ACs which are theoretically consistent – i.e. increasing in emission ceilings – and plausible from an engineering and economic viewpoint.
3. Abatement strategies depend on farm characteristics and chosen indicator.
4. Indicators show different economic efficiency based on their normalized MACs.

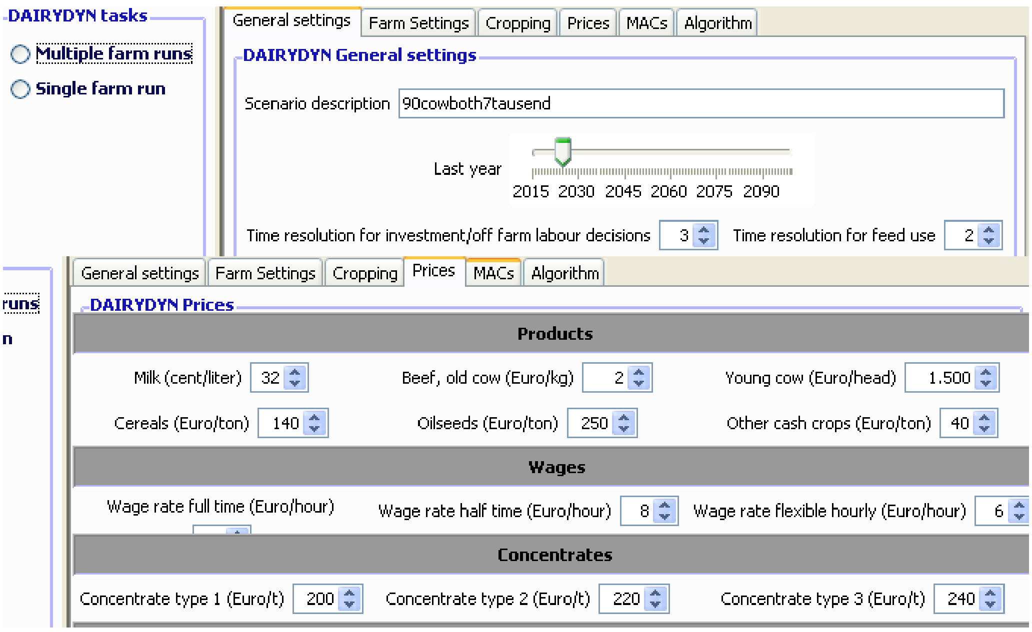
5.6 Technical implementation

The model template is realized in the *General Algebraic Modelling System* GAMS (ROSENTHAL, 2010). It is complemented by the so-called coefficient generator, i.e. GAMS code, which parameterizes an instance of the model template based on bio-physical relations (such as requirement functions for animals) and engineering data (such as look-up tables with investment and other costs and labor requirements per stable place and year for different stable types). The coefficient generator is designed to be generic enough to cover relevant dairy farm types in Germany and to define all necessary model parameters from a few, decisive initial farm characteristics such as given herd size and milk yield, land, labor and stable endowments.

Based on the current, not yet fully developed template, a typical application for one farm over a planning horizon of 15 years leads to a MIP problem with about 20 thousand variables of which about 400 are integer. An efficient MIP solver combined with an efficient solution strategy to handle the step-wise GHG reduction is hence needed to keep overall solution time manageable. We opted to apply CPLEX 13.2 (IBM, 2011) in parallel solving mode combined with automatic tuning, using integer re-starts from previous solves and MIP solution tolerances derived from the objective value in the reference and solving on a performing 8 core computing server. Equally, in order to reduce model size, some decision variables in the model relate to several years and re-investment are only possible at specific time point and not in each year. These settings can be changed in sensitivity experiments to verify that they have a serious impact on results.

Solving a single model instance for one indicator and emission ceilings with a 15 years planning horizon takes between 10 and 60 seconds. Accordingly, a run to simulate MACs for four indicators and twenty reduction steps easily can take as long as 60 minutes.

Figure 3: **Sections of the graphical user interface**



Source: own illustration.

A Java based Graphical User Interface[39] (GUI, see figure 3) allows defining the farm types, generating an instance of the template model, its application on a set of indicators and GHG reduction steps and result analysis based on tables and graphs.

5.7 Illustrative application

For the first step, different dairy farm types (differentiated e.g. in starting size and milk yield potential) are simulated under the four different emission indicators discussed above to show the impact of indicators on the differences in costs to abate emissions and to underline the indicator-dependent choice of abatement options as well as the differences concerning the accuracy of different indicators.

Main characteristics of the modeled farms

For our illustrative experiments, we simulate four farms differentiated by initial herd size (60 or 90 cows) and cow milk yield in the first simulation year (5000 or 7000 kg per cow and year).

Because of the bigger initial herd size, the 90 cow farms are endowed with a family work force of 2 instead of 1.5 annual labor units, further on, it possesses more land and benefits from lower labor need per animal compared to 60 cows farm. The planning and thus optimization horizon is assumed to end in the year 2025 with a construction year of the stables in 1995 (adapted to the assumed useful live of 30 years for buildings). The average price for milk is fixed at 0.32 €/kg. The runs encompass three states of nature: one with average prices, one with 20% higher prices for animal products and one with an increase in crop and concentrate prices by 20%. Abatement options depend on the chosen indicators as discussed above. The analysis is complemented by a sensitivity analysis for how manure application is handled. In the standard model, the farm spreads manure with own equipment so that switching the application technique requires investments. In our sensitivity experiment, we let the farm use contract work instead: that leads to somewhat higher per unit costs if the equipment would be fully depreciated over the planning horizon (which does not happen in our experiments), but gives the farm more flexibility.

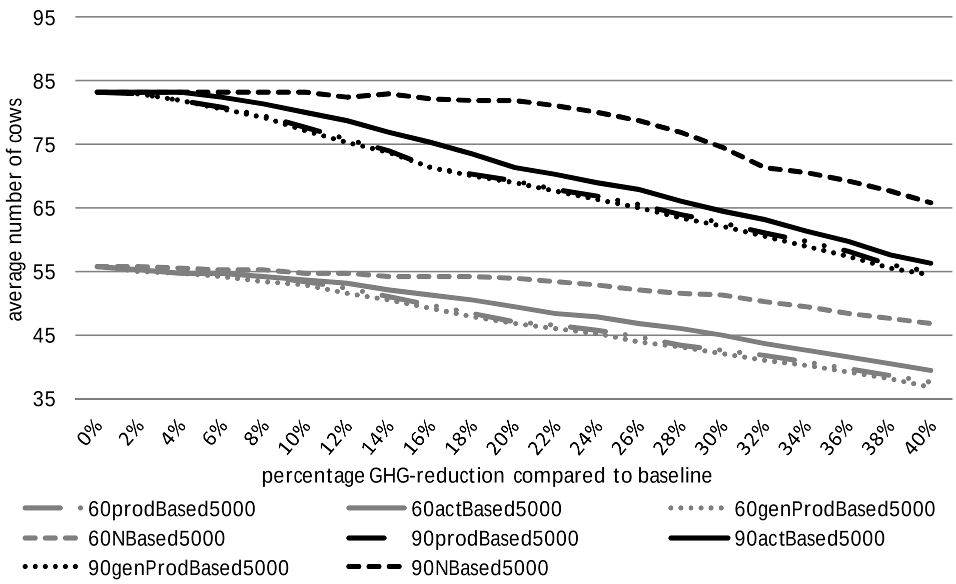
[39] The exploitation part draws on the CAPRI Graphical User Interface (BRITZ, 2011).

5.8 Results

Herd sizes

The following figure 4 visualizes the average herd sizes over the whole planning horizon, under different GHG reduction levels for the case of 60 and the 90 cows initial herd size and an identical initial milk yield of 5000 kg head^{-1} year^{-1}.

Figure 4: **Average herd size over planning horizon for different GHG reduction levels**



Source: own calculation and illustration.

Note that in base run, the farm will typically towards the end of the simulation horizon reduce its herd to avoid raising calves and heifers to replace cows. The herd is sold in the last year at an assumed relatively low price which is below the endogenous replacement cost if cows are not used the full number of lactations. That explains why average herd sizes are somewhat below the initial ones.

The graphic highlights that herd size reductions differ strongly between indicators, but that relative reductions between the 60 and 90 cow farms are quite similar. The largest reductions are found under the prodbased and genProdBased indicators, followed by the actBased indicators whereas the NBased indicator requires the smallest herd size adjustments.

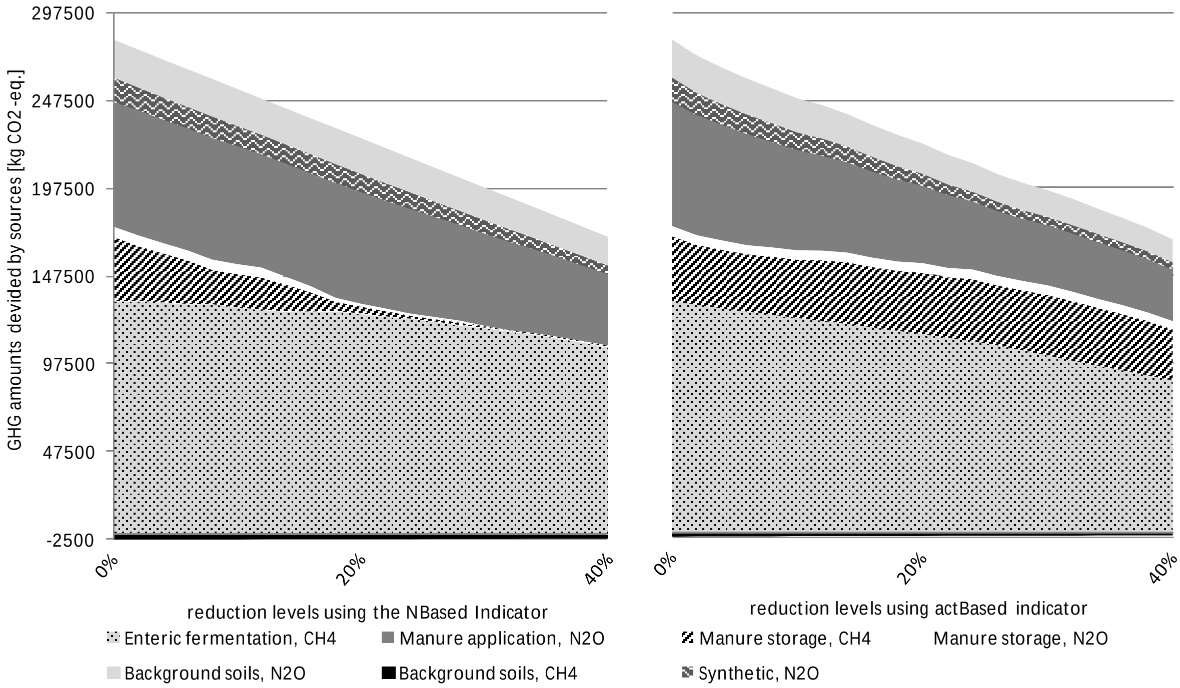
The sharper reduction under the production based indicators look at first glance astonishing, as the emissions per cow are higher under the activity based indicator for a 5000 liter cow. For the production based indicator, the default emissions per cow of ca. 3300 kg CO_2 equivalents under the activity based indicator are converted assuming a milk yield of 6000 l. Accordingly, a 5000 liter cow will emit only around 2750 kg CO_2 equivalents under the production based indicator (compare table 1 above). So why does the farm need to reduce its herd size more under the indicator prodBased with the lower emissions per cow? The reasons are twofold. Firstly, abatement efforts of the farms are defined relative to the indicator. So while indeed total accounted emissions under the prodBased indicator are lower, the relative reduction required is the same. And secondly, linked to that reason, due to lower emissions per cow under the production based indicator, the share of emissions from crops in the baseline is higher compared to the actBased indicator. Emissions from crops are more expensive to abate under that simple indicator, as their reduction requires giving up own fodder production and replace it by concentrates. The GHG emissions linked to concentrate production (e.g. fertilizing of cereals or oilseeds used for cake production and related background emissions from soils) would be accounted in other farms or even other countries, underlining again the importance of the system boundary definition.

The NBased indicator affects herd sizes only at higher reduction levels as cheaper abatement possibilities such as changing the manure storage type are used which are not accounted for by the other indicators. That allows abating 40% of the initial GHGs with herd size adjustments of -16% (60 cows) resp. -20.9% (90 cows), whereas the other indicators require reductions between -29.4% and -34.6%, depending on the indicator and herd size.

Abatement strategies under different indicators

Figure 5 highlights differences in abatement strategies between the NBased and a simpler one, the actBased indicator, using results for the farm with an initial herd size of 60 cows and 5000 liter as an example. The graphic shows cumulated source specific emissions (expressed in CO_2-equivalents) based on the accounting rules of the NBased indicator. The reader is reminded that emissions under the activity based indicator are however reduced according to default emission factors attached to herds and crop hectares found in table 1.

Figure 5: **GHG by sources for 60 initial cows with 5000 kg yield level, emission restrictions based on actBased and NBased indicator**



Source: own calculation and illustration.

The chart on the right hand side illustrates GHGs emitted from different sources when the farm has to abate according to the activity based indicator. It first underlines that enteric fermentation and manure application are the two dominating sources of emissions in our example farm. One can clearly see that there is an almost linear reduction of almost all sources under the activity based indicator. The decrease in CH_4 from enteric fermentation (34% reduction compared to baseline) is linked to the reduction of the herd size, whereas emissions from application of manure and synthetic fertilizers as well as background emission from soils are driven by a proportional reduction in land use: the farms rents out the hectares which are not longer used for fodder production as it seems not economically attractive to change the feed composition per cow (grassland under cultivation lowered by 40%). Indeed, the only exemptions from the linear reduction are emissions stemming from manure storage which are rather constant in case of the actBased indicator. Obviously, the existing manure storage is a binding constraint, but an expansion by new investments too expensive.

Contrary to the farm management under the actBased constrained farm, the left hand side of figure 5 illustrates the fundamentally different abatement path under the NBased indicator. Up to about 18% reductions in GHGs, the farm almost entirely abates via reduction of GHGs from manure storage: it first uses straw cover and latter the far more expensive foil coverage to reduce CH_4 and N_2O emissions from the slurry tank. Beyond that point, the abatement strategy is almost equal to the one under the activity based indicator: herd sizes are reduced accompanied by a proportional adjustment in land use. The reduction from manure management is by far stronger than the herd size adjustments: higher N_2O emissions from manure applied to pasture allow reductions by switching from grazing to mowing.

A perhaps astonishing finding is the fact that enteric fermentation is reduced more than the lower dairy herd suggests. That is linked to the fact that the farm has to abate GHGs in average over the planning horizon. By reducing the herd size much stronger towards the end of the planning horizon, it can achieve an over-proportional reduction in replacement needs. For higher reduction levels, no heifers are kept for the last 4-5 years and cows leave the herd after their maximal number of lactations without being replaced.

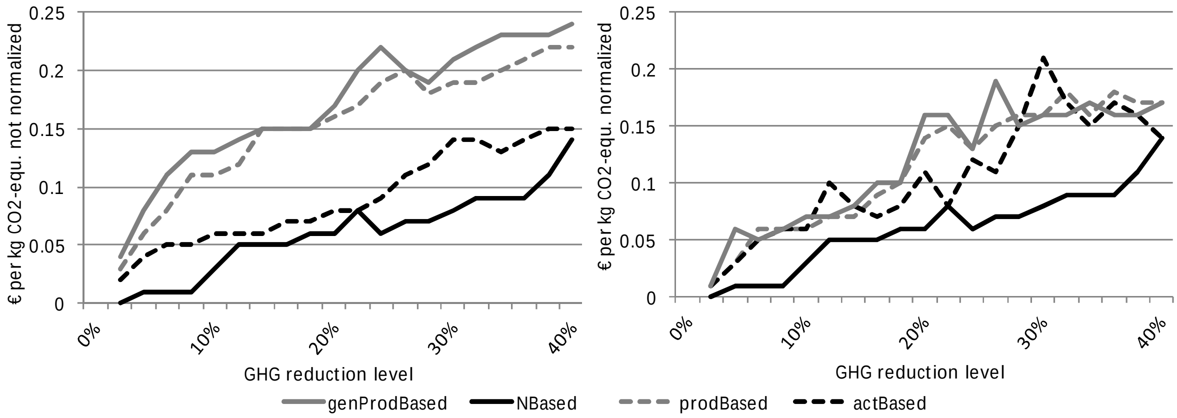
The results hence underline that abatement strategy are clearly depending on the indicator. Thus, despite almost identical GHGs abated (both reduce from about 280 t to 167 t CO_2-equ. year^{-1}) when measured with the more accurate NBased indicator, significant differences in abatement costs can be expected between the indicators.

MACs under different indicators

Either way, achieving abatement of GHG emissions will cause costs on farm level or reduce overall profits of the farm as GHG ceilings acts as restrictions. Figure 6 shows the MAC curves under the different emission indicators. The left hand side shows the MACs which drive the abatement strategies, i.e. the ones under the indicator used to define the emission ceiling (see equation 2.1. for the definition). As to be expected, the NBased indicator creates the lowest MACs for all reduction steps by offering the largest sets of abatement possibilities. The abatement cost for one unit of additional emission abatement range from 0 to 140 €/t CO_2-equ. The actBased MACs are much higher up to a reduction level of around 8% compared to the NBased one, which abates in that range by using straw cover on the slurry tank which is rather cheap. The two curves then come closer as the NBased indicator is switching to foil cover, which is far more expensive.

Figure 6 underlines that the not normalized marginal abatement costs for the prodBased and genProdBased indicator are almost identical, but generally much higher compared to the actBased and NBased MAC curves. As mentioned above, the 5000 l cow receives a kind of discount under the actBased indicator as the emitted GHGs per animal are overestimated.

Figure 6: **Normalized and not normalized MAC curves for 60 cows initial herd, 5000 kg milk head^{-1} year^{-1} initial yield potential [€/kg CO_2-equ.]**



Source: own calculation and illustration.

These MAC curves on the left hand side of figure 6 are the relevant ones for decisions at farm level as they drive the abatement strategies. But the simpler indicators might over- or underestimate the real abated GHGs compared to the more complex and accurate NBased one, and consequently, also provide biased results for the profit foregone per "real" GHGs abated. That can be clearly seen from the right hand side where the normalized abatement costs are shown according to equation 2.2.

The NBased indicator as our most accurate accounting scheme is used as the reference indicator and for normalization of MACs (equation 2.2). Hence, the normalized MAC curve for the NBased indicator is identical to the one on the left hand side. Comparing the normalized MACs on the left hand side and the ones on the right hand side shows if the indicators account for more or less GHG abated in relation to the indicator used for normalization. Imagine we used the prodBased indicator to steer abatement effort of farmers, but know that the true GHGs relevant for the climate warming effect of dairy farms can be measured with the NBased indicator. The curves suggest that if farmers abate a certain percentage of GHGs measured by the prodBased indicator, they have effectively abated less "true" GHGs. So in order to judge how expensive it was to abate the GHGs from a public good perspective, we relate the "true" change in the externality to the costs faced by the farmers. Thus, if the normalized MACs are higher than the not normalized ones, the indicator scheme overestimates GHG reductions and underestimates the real abatement costs and vice versa.

The first point to note is that the two production based indicators overestimate the "true" abatement costs, i.e. the farms abate in reality more GHGs than the indicator, used to define the emission ceilings, suggests. The opposite effect is found in case of the actBased indicator for wider parts of its normalized MAC curve: the "discount" in form of higher emissions per cow leads to overestimation of the abated GHGs.

We conclude that the normalization of the MAC curves of different indicators is necessary to draw correct conclusion regarding indicator recommendations. The not normalized MACs of the genProdBased and prodBased indicators signal high marginal costs at all reduction levels and would suggest implementing rather the actBased indicator which has the additional advantage of being simpler. The normalization shows however that the actBased indicator overestimates the abated GHGs and is economically less effective. But nevertheless, both types of MACs are important for analysis: The not normalized ones show the profit losses incurred to farms by imposing an emission ceiling based on a specific indicator. The normalized MAC curves are relevant from a societal point of view to check if the indicator sends the right economic signals to the agents when GHGs are accounted based on the best available indicator.

MACs depending on farm attributes

Finally, we turn our attention to the question to what extent farm characteristics such as size or milk yield impact abatement costs, using the normalized MACs. In order to show the effect of farm attributes on the abatement costs, the profit loss for the total reduction of 40% is divided by the related reduction in GHGs when measured with the NBased indicator to derive average normalized mitigation costs per kg of CO_2-equivalent as shown in table 2.

Table 2: **Average normalized abatement costs by farm characteristics and emission indicator [€ per kg CO_2-equ.]**

Initial herd		actBased	prodBased	genProdBased	NBased
60 cows	5000 liter	0.10	0.11	0.11	0.06
	7000 liter	0.14	0.15	0.15	0.08
90 cows	5000 liter	0.12	0.13	0.13	0.09
	7000 liter	0.09	0.10	0.10	0.05

Source: own calculation and illustration.

It is obvious that all three simpler indicators lead to much higher average abatement costs compared to the NBased one. A 90 cow farm with 7000 liter cows could almost halve the abatement costs if the NBased instead of the actBased indicator is used.

A marked result is that the NBased indicator induces the lowest average abatement costs per kg CO_2-equ. (always below 100€/t) independent of farm characteristics. But for all indicators, average abatement costs per kg differ depending on the starting herd size as well as on intensity level.

Sensitivity experiment for manure handling

A sensitivity analysis was done for the 60 cow farm with initial yield level of 5000 kg in order to highlight the effect of sunk costs on the abatement strategy. In the runs depicted above, it is assumed that the farm owned already a simple manure barrel; a switch to other application techniques would require additional investments. The sensitivity analysis is based on an alternative assumption: manure spreading is based on contract work, allowing to flexibly switching between application types.

Table 3: **Share of manure by application type, contract work compared to investments in application machinery**

		GHG reduction level [%]				
		0	10	20	30	40
contract work	broad spread	2.0%	2.5%	25.1%	39.3%	76.6%
	drag hose	98.0%	97.5%	74.9%	60.7%	23.4%
investment	broad spread	100.0%	100.0%	100.0%	100.0%	100.0%

Source: own calculation and illustration.

Under the NBased indicator which is the only one accounting for changes in application techniques, distinct differences in manure application management are noticeable and shown in table 3.

Under the sunk cost case, new investments in an injector or a drag hose are always too expensive and manure is always broad spread. If manure spreading is based on contract work, the farms will in the baseline use the drag hose option: it reduces ammonia losses and thus saves synthetic fertilizer. The injector option would reduce losses further, but is too expensive. Under the GHG emission ceiling, it is cheaper to waste some N as ammonia instead of carrying abatement costs linked to higher N_2O losses when manure instead of synthetic fertilizer is used on pasture.

Conclusions from result section

Based on these illustrative results, preliminary statements can be made concerning the hypotheses formulated at the end of section 5.5. As clearly shown above by figure 6 and table 2, the shape and level of MAC curves depend on the initial farm characteristics as well as on the chosen GHG indicator. Furthermore, the MACs increase in abatement levels, which is plausible provided that the decision maker always chooses the next cost efficient abatement option. Consequently, the overall abatement costs (ACs) rise with higher emission reductions as well. The abatement costs are within the range of results from other studies. DECARA et al. (2005: p.566) derive maximum MACs of 20 €/t CO_2-equ for different European farm types under a 3.9% emission reduction, our results for a reduction level of 4% to baseline lead to marginal abatement costs of 10 to 60 €/t depending on the chosen indicator scheme for the example farms. PÉREZ and BRITZ (2003) come up with average marginal abatement costs of 53 €/t for an EU wide 10% reduction of agricultural emissions using the CAPRI modeling system. Our model derives MACs between 30 and 60€/t for a ten percent GHG reduction (cf. figure 6). Hence, the above stated model results are within ranges of scientific findings from other studies. However, the reader should keep in mind that studies mentioned above derive costs for larger farm aggregates whereas our results only represent single example farms.

The results also underline that abatement strategies depend on the indicator as shown in figure 4 and discussed based on the emission sources shown in figure 5. With regard to the economic efficiency of different indicators and abatement strategies based on the normalized MACs (described in section 5.5), the NBased indicator (here taken as the reference indicator) shows the highest level of economic efficiency in abating GHGs from dairy farms. It however also requires measuring and controlling e.g. manure application quantities by spreading technique which might be expensive or even impossible.

Our sensitivity experiment underlines the importance of sunk costs for the abatement strategies and motivates the application of a dynamic simulation framework over a longer optimization horizon to capture investment based options additionally to more flexible mitigation possibilities.

Expected results after model completion

Further steps will complete and expand the model template, apply it to much more farms, expand the planning horizon, perform sensitivity analyses and finally, derive aggregate results for German dairy farms. Especially the addition of a more elaborate list of GHG abatement options (e.g. feed additives, changes in feed digestibility) will refine the analysis regarding the normalization of GHGs and might help to find economic effective abatement strategies. Statistical analyses will reveal the relation between farm attributes and MACs, and help to derive aggregate regional and sector wide MAC curves.

Policy conclusions

The still illustrative applications do not yet allow for immediate policy recommendations. But even the preliminary results underline the key role of an appropriate indicator choice:

marginal abatement costs differ considerably between indicators while differences between GHGs estimated based on “state of the art” calculations and those estimated with simpler indicators can be substantial. The notion of “better” for an indicator has at least three interlinked dimensions: (1) the accuracy in measuring emissions, (2) its ability to trigger cost minimal abatement strategies, and (3) the implementation and monitoring costs (not discussed above).

There is clearly more analysis needed which also takes monitoring costs and the administrative burden for farmers into account. Assume, to use a hypothetical example, that analyses would reveal that strategies under a complex, very hard to actually implement and control indicator with low MACs do not differ across farms. All farmers would choose the same easy to observe strategy such as an investment in foil silo coverage to reduce GHGs in a cost efficient way. One might conclude that the most efficient policy is to enforce the strategy on all farms rather than to implement economic instruments based on a GHG indicator scheme which would only lead to additional private and public costs related to its implementation on each farm on top of the actual abatement costs.

5.9 Conclusion

The paper discussed the structure and application of a farm-specific economic simulation model for German dairy farms which is able to cover a great variety of GHG abatement options and to derive farm specific marginal abatement cost curves for different emission indicators. We argued that a fully dynamic model integrating binary and integer variables is necessary to analyze GHG abatements in dairy farms. Illustrative model results showed that the model template creates robust and economically reasonable reactions to emission ceilings, that the choice of emission indicator has a significant impact on abatement costs and that abatement strategies as well as MACs depend on farm attributes such as herd size or milk yield. Our findings underline that the choice of emission indicator is indeed a core question in environmental policy design as simpler, more aggregate indicator schemes can lead to quite biased results.

Further research is necessary to improve the indicators, include more abatement options in the model template and apply it systematically to farms with different attributes to allow scaling up to sector level. Equally, a final evaluation of indicators will require taking also control and implementation costs into account.

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Chapter 6: **A comparison of emission calculations using different modeled indicators with 1-year online measurements**[40]

Abstract

The overall measurement of farm-level greenhouse gas (GHG) emissions in dairy production is not feasible, from either an engineering or administrative point of view. Instead, computational model systems are used to generate emission inventories, demanding a validation by measurement data. This paper tests the GHG calculation of the dairy farm-level optimization model DAIRYDYN, including methane (CH_4) from enteric fermentation and managed manure. The model involves four emission calculation procedures (indicators), differing in the aggregation level of relevant input variables. The corresponding emission factors used by the indicators range from default per cow (activity level) emissions up to emission factors based on feed intake, manure amount and milk production intensity. For validation of the CH_4 accounting of the model one-year CH_4 measurements of an experimental free-stall dairy farm in Germany are compared to model simulation results. An advantage of this interdisciplinary study is given by the correspondence of the model parameterization and simulation horizon with the experimental farm's characteristics and measurement period. The results clarify that modeled emission inventories (2,898, 4,637, 4,247, 3,600 kg CO_2-equ. cow^{-1} $year^{-1}$) lead to more or less good approximations of online measurements (av. 3,845 kg CO_2-equ. cow^{-1} $year^{-1}$ (±275 owing to manure management)) depending on the indicator utilized. The more farm-specific characteristics are used by the GHG indicator; the lower is the bias of the modeled emissions. Results underline that an accurate emission calculation procedure should capture differences in energy intake, owing to milk production intensity as well as manure storage time. Despite the differences between indicator estimates, the deviation of modeled GHGs using detailed indicators in DAIRYDYN from on-farm measurements is relatively low (between -6.4 and 10.5%), compared with findings from the literature.

Keywords: *agricultural modeling, GHG measurement, validity of modeled GHGs, emission indicators, dairy farm methane emissions, DAIRYDYN, enteric fermentation*

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[40] This part is published as LENGERS, B., SCHIEFLER, I. and W. BÜSCHER (2013): A comparison of emission calculations using different modeled indicators with 1-year online measurements. *Environmental Monitoring and Assessment* 185(12):9751-9762. The final publication is available at link.springer.com. Bernd Lengers contributed to the whole document except of section 6.2.2. The development of the DAIRYDYN model was funded by a grant from the German Science Foundation (DFG), reference number HO 3780/2-1. The authors are grateful for the cooperation of the Chamber of Agriculture of North-Rhine Westphalia, where the measurements were carried out. This investigation was funded by the Landwirtschaftliche Rentenbank and the Federal Ministry of Food Agriculture and Consumer Protection, Germany.

6.1 Introduction

Greenhouse gases from agricultural production systems are discussed broadly in a scientific as well as a public and political context. As mentioned by the Food and Agricultural Organization (FAO), in 2004 and 2007, dairy production systems in particular, supposedly bore a large part of global agricultural livestock GHG inventories (ca. 16%), and about 2.7% of global total anthropogenic GHGs (FAO, 2010: p.32; HAGEMANN et al., 2012; STEINFELD et al., 2006: p.96-112). However, real measurements of emissions are not realizable for a large number of farms, or even whole regions. Many methods and schemes have been designed to calculate GHG emissions from arable production systems and animal husbandry, while only knowing some farm- or regional-specific data on different aggregation levels. Implemented into specific model approaches — for example, RAINS (ALCAMO et al., 1990); EFEM (KAZENWADEL and DOLUSCHITZ, 1998); MDSM (LOVETT et al., 2006); a study by HAGEMANN et al. (2012), based on methane equations from KIRCHGESSNER et al. (1991); or a model approach used by DECARA and JAYET (2000), which calculates GHG inventories from specified regions in the European context — the available information led to modeled GHG estimates. Others also developed single-farm approaches for predefined single-farm types. For instance, SCHILS et al. (2007) used the single farm model DairyWise for their estimations and WEISKE et al. (2006) presented results using a farm GHG model which was originally developed by OLESEN et al. (2004)).

Since the modeled GHG emissions have to be seen as a proxy for the actual GHG emissions of the modeled real-world systems, the question arises if the validity of computational models is given on a sufficiently high level. This topic has already been discussed by BURTON and OBEL (1995), depicting the balance of model realism, and the overall purpose of the modeling approach. The inherent model functions are not able to show real ongoing biochemical or bio-economic processes precisely. For instance, there are assumptions and simplifications, and also not yet full understanding of biochemical processes e.g. in the rumen (STORM et al., 2012). However, the results should, nevertheless, display an adequate proxy for outputs of the real-world system. But as the predictive character of a model can only be '[...] as good as the accuracy of the mathematical method or equations [...]' (ELLIS et al., 2010: p.347), it is quite difficult to build up a consistent model approach for GHG release from complex production systems (HERRERO et al., 2011). Hence, depending on the specific definition of emission calculation procedures, different accounting biases concerning the GHG inventories may occur.

Validation of GHG modeling is done mostly by using small-scale and/or short-term measurements (respiration chambers, indirect calorimetry, mass balance, hood calorimetry). ELLIS et al. (2010) for example used such data for the validation of nine different ruminant dairy CH_4 equations and MILLS et al. (2001) applied it for validation of their modeling of methanogenesis in a lactating cow. Only TALLEC and HENSEN (2011), up to now, have compared modeled and measured CH_4 estimates over a longer time period of more than a few days duration (one-month field experiments) from dairy livestock on grassland, by using a simple Gaussian plume model formerly developed by HENSEN and SCHARFF (2001). However, as also criticized by the authors themselves, measurements over one month are not sufficient for accurate validation results. For our purposes, there are few published CH_4 emission factors from modern dairy free-stalls with a slatted floor: e.g. KÜLLING et al. (2002), SCHNEIDER et al. (2006), SNELL et al. (2003) and ZHANG et al. (2005).

However, the published data stem mostly from short-term measurement intervals (from 2–3 days per season (SNELL et al., 2003) to several weeks (SCHNEIDER et al., 2006)). Other data, based on individual animal measurements, are often restricted to a limited number of animals, and/or do not include emissions from managed manure (e.g. respiration chambers (DERNO et al., 2009)). Hence, the estimates may be biased by not being able to cover seasonal and yearly external or internal variability in the production process, when extrapolating the derived per day emission factors to default one-year emission parameters, per animal, or per livestock unit (LU) (one LU is equivalent to, for example, a cow with a live weight of 500kg). The comparability of literature estimates is especially hindered with regard to the differing cattle breeds, milk output intensities and present lactation phase of the animal population investigated in the studies. Additionally, the above mentioned studies offer highly varying CH_4 emission factors per LU and year, ranging between 2,221.8 kg CO_2-equ. and 4,063.9 kg CO_2-equ. (ZHANG et al., 2005), and hence would lead to imprecise validation of emission simulations when applying these as reference. Owing to a lack of production-specific information about the experimental units underlying these studies, one is not even able to adjust parameters in a farm-level model approach for equivalent circumstances, which would perhaps increase the usability of the literature findings for validation purposes. Furthermore, small-scale measurement results are regarded as not being appropriate for comparison with long-term calculations for high animal numbers (STORM et al., 2012: p.175).

The problem of obtaining reliable data for validation is also of relevance for the simulation of GHG emissions by the bio-economic dairy farm-level model DAIRYDYN (LENGERS and BRITZ, 2012), for specialized dairy farms on slatted floors. The model allows for choosing one out of four different emission-calculation schemes (indicators), and accessing more or fewer aggregated system variables of the dairy production process (e.g. default emission factors per activity or precisely connected to feed intake). LENGERS and BRITZ (2012) applied the approach to analyze the effect of GHG accounting on chosen abatement measures and adherent mitigation costs, if farms are restricted by emission ceilings.

The objective of this study is to test the accuracy of CH_4 calculation by different designed GHG calculation schemes for lactating cows and stored manure of the DAIRYDYN model. Therefore, we apply the model approach with adherent GHG calculation procedures on a real existing dairy stable complex. Modeling results are compared with results from experimental measurements in a free-stall dairy barn in Germany (Haus Riswick). The experiments are characterized by long-term measurements over one year, covering seasonal variations, and thus result in more precise values than emission factors based on projections with only a few measurement days (in contrast to ELLIS et al. (2010)). For biological processes, long-term estimation horizons are particularly important. Recent studies have shown that there is a significant variation of individual CH_4 emissions between single cows (278 to 456 g CH_4 day^{-1}; GARNSWORTHY et al., 2012), whereby the number of animals investigated may play an important role in the measurement accuracy.

Furthermore, the own measurements include emissions from animals' release, as well as emissions from liquid manure, hence reflecting all sources of emissions from the dairy barn. Since the quantification of GHG emissions at barn level (sum of animal and manure) is studied less thoroughly, this is a clear advantage over some other studies, which may be limited to the animals' release (e.g. static respiration chambers), and only measure small livestock numbers (JOHNSON et al., 1994: p.361; MOE and TYRRELL, 2010).

To follow the objective, the computational modeling approach, used by the DAIRYDYN model, will be explained; in particular, concerning the different emission calculation schemes which can be chosen by the user. Afterwards, the experimental set-up of the dairy barn on Haus Riswick will be explained, focusing briefly on the measurement approach. The implementation of specific farm characteristics of the dairy free-stall on Haus Riswick into DAIRYDYN will allow for simulation and comparison of an equivalent model farm and adherent CH_4 release. The modeled and the measured data cover the same time period with a high representative animal number. This will improve the validation of model calculations by more reliable results, because seasonal and farm exogenous aspects are also captured by the measurements.

6.2 Material and methods

6.2.1 Model concept of DAIRYDYN

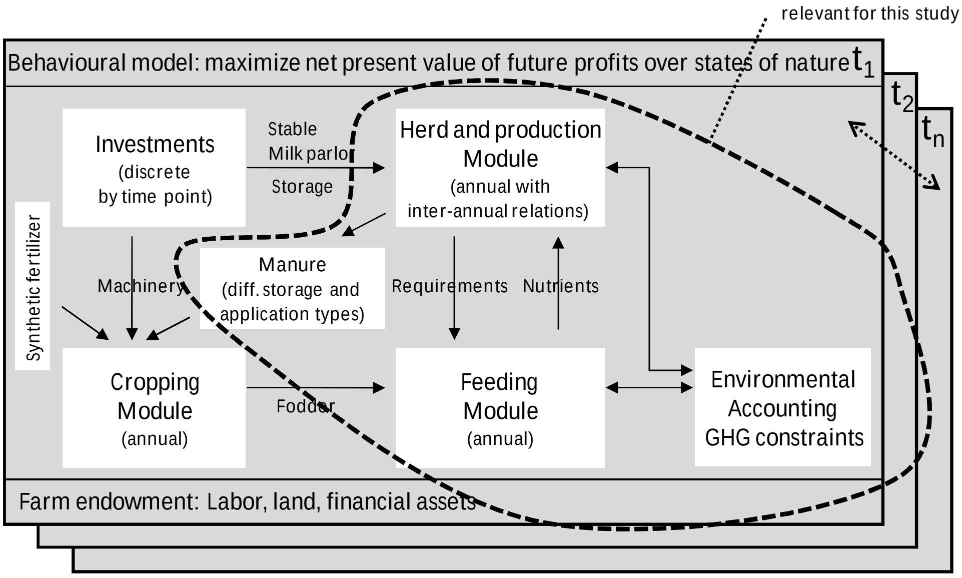
The DAIRYDYN model is a farm-level model developed by LENGERS and BRITZ (2012), with an objective function of maximizing net present value of future profits, using different natural states. DAIRYDYN was built for the process-based modeling of single dairy farm development, *inter alia* the occurring GHG emissions combined with the production process. Therefore, the model user can choose from four different emission calculation schemes, based on consistency-proven IPCC (Intergovernmental Panel on Climate Change) methodology with several enhancements.

The model uses a fully dynamic mixed integer linear programming approach. It is programmed with the general algebraic modeling system GAMS, using the industrial solver CPLEX (IBM, 2011). It enables the user to simulate farm-level development of specialized dairy farms (including calves, heifers and acreage) over various planning horizons. Animals are differentiated concerning milk yield potential, lactation number, as well as lactation phase. Feeding rations can be changed quarterly, whereby self-produced ground-bait can be supplemented by different concentrates. Manure excretion rates and adherent nitrogen amounts are also captured on a monthly basis. Beneath the baseline farm development, management and cost implications through farm-level emission ceilings can be analyzed, deriving GHG-indicator-specific marginal abatement cost for GHG mitigation efforts at the single-farm level.

Figure 1 shows bio-economic interactions between the modules that are implemented into the used model approach. The inherent emission calculation rules (indicators) quantify production-specific GHG inventories. Emission calculations are related to source (manure management, enteric fermentation, arable production, etc.) and gas type (CH_4, N_2O).

The measurements on Haus Riswick were limited to the barn including manure storage and did not include emissions from e.g. crop production, fertilizers or machines (see section 6.2.2). Hence, only those modules within the dotted line are of relevance for the following model calculations (Fig 1).

Figure 1: **Overview of DAIRYDYN model and relevant modules**



Source: following L*ENGERS and* B*RITZ (2012: p.123).*

Emission indicators for GHG modeling

As noted above, different emission calculation schemes can be chosen by the model user. The four calculation schemes differ in the detail of farm specific production variables that are relevant for the calculation. For instance, emissions can be calculated in a very simplified way only using parameters of the principle activity (herd size and cropping ha). To go one step further, more detailed parameters, like mass flow and feed composition can be included in the calculation.

A detailed description of the indicator schemes is given by a former study of LENGERS and BRITZ (2012). The simplest indicator is the activity-based one *(actBased).* It multiplies default emission factors per head or per ha (taken from IPCC (2006) Tier 1 level) with activity levels to derive whole-farm emissions. The production-based *(prodBased)* indicator differs in calculation of emissions from cows and crops. Therefore, the *prodBased* indicator is implementing static emission factors per unit of product (e.g. per kg of milk output). These emission factors are derived from the default Tier 1 values (emission parameter for milk is derived by dividing IPCC Tier 1 default factor by an assumed average milk yield per cow per year of 6,000 kg). However, the default per unit of product emission factors lead to various overall emissions depending on per ha or per stable place output level as it suggests a linear increase in CH_4 release per cow or per ha with increasing output. The *genProdBased* indicator also recognizes the diminishing emissions per kg of milk, when intensity level of cows increases (emissions from gross energy intake for maintenance and activity are allocated to higher milk output), assuming decreasing emissions per kg milk with increasing milk yield per cow and year (derived by Tier 2 approach with standard energy digestibility of 60% (IPCC, 2006)). Manure is assumed to be stored for half a year on average. A more detailed emission calculation is presented by the *NBased* indicator, recognizing single animal gross energy demand for animal emission calculation, depending on the actual lactation phase, and with adjusted average feed digestibility for real circumstances. Furthermore, it uses monthly manure amounts in storage to calculate emissions by different manure management types (subfloor, surface storage, coverage techniques). Emissions stemming from arable production processes are based on N application (synthetic and organic). Emissions from storage and arable N application are implemented on a monthly basis, to capture effects of manure removal and application frequency as well. The CH_4 calculation formulas, implemented into the model to derive emissions from lactating cows and sub-floor stored manure, are shown in Table 1.

Table 1: **Methane production equations relevant to the investigated farm unit**

		Equations*			
Indicator	Unit	Enteric fermentation	Manure storage	Comments	Source
actBased	CH_4 (kg cow^{-1} year^{-1})	117	21	default values	IPCC (2006) Tier 1
prodBased	CH_4 (kg cow^{-1} year^{-1})	$117/6000\ liter \times milkyield\ (liter\ cow^{-1})$	$21/6000\ liter \times milkyield\ (liter\ cow^{-1})$	linear increase per output unit	IPCC (2006) Tier 1
genProdBased	CH_4 (kg cow^{-1} year^{-1})	$GE_l(MJ\ year^{-1}) \times Y_m/100/55.65$	$VS(m^3 year^{-1}) \times B_0 \times 0.67 \times MCF/2$	half year manure storage assumed & default energy digestibility of 60%	IPCC (2006) Tier 2
NBased	CH_4 (kg cow^{-1} year^{-1})	$\sum_p GE_{pl}(MJ\ phase^{-1}) \times Y_m/100/55.65$	$\sum_m VS_m(m^3 month^{-1}) \times B_0 \times 0.67 \times MCF/5.66$	monthly storage emissions & experiment adjusted digestibility	IPCC (2006) Tier 2

*selection of equation relevant default parameters in line with IPCC (2006) methodology for Western Europe.
GE_l= one year gross energy demand for cow with specific milk yield level l; GE_{pl} = gross energy demand for specific phase of lactation p and milk output potential l of each cow; Y_m = methane conversion factor (6.5% of GE in feed converted to methane); VS = volatile solid excretion cow^{-1} year^{-1} on a dry-organic matter basis; B_0 = maximum methane production capacity for manure (m^3 CH_4 kg^{-1} of VS); *0.67* = conversion factor of m^3 CH_4 to kg CH_4; MCF = one year methane conversion factor for sub-floor manure storage; VS_m = monthly VS in sub-floor pit.

Source: own illustration following IPCC 2006 and LENGERS and BRITZ (2012).

Table 1 gives a systematic view of the CH_4 calculation concepts of the four applied indicator schemes, for CH_4 release from enteric fermentation as well as stored manure amounts, presenting a growing level of detail from top to bottom.

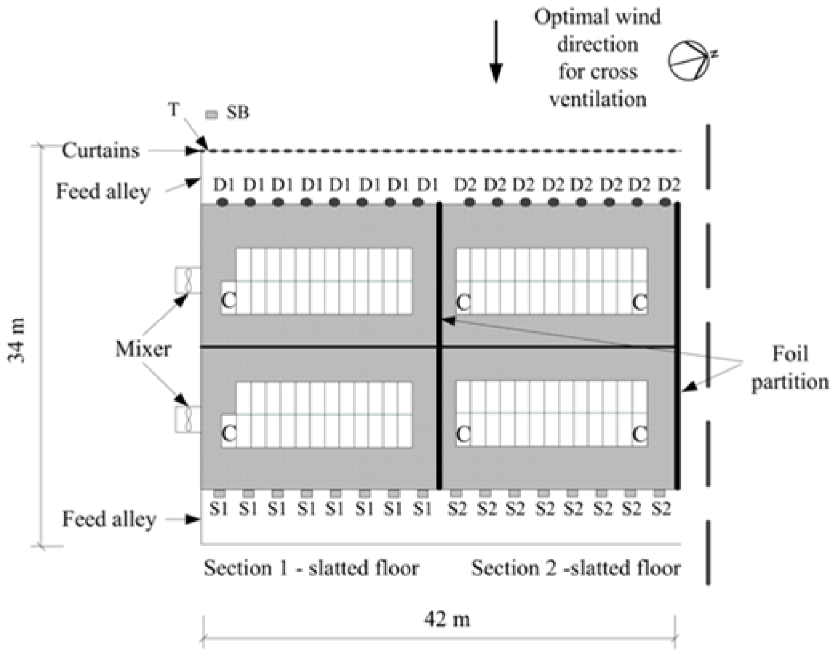
6.2.2 Measurement installation on Haus Riswick

6.2.2.1 Site description

Measurements used for model validation were carried out in a newly built dairy barn of the Chamber of Agriculture North-Rhine Westphalia, at the Centre of Agriculture, Haus Riswick, in North-Western Germany. The annual average temperature of the investigated site was 9.8 °C (see Fig. 2, measured at feed alley with open curtains). The average outdoor temperature ranged from 4.3°C monthly mean in January 2011 up to 18.6°C monthly mean in August. Mean humidity in 2011 was 79%, mean wind speed was 1.9 m s^{-1} and the main wind directions in 2011 were South-West and West (data from nearest official weather station in Goch). The dairy cows were kept in a free-stall dairy barn with an external milking parlor, during the whole year. Two equal-sized compartments (section 1, section 2) of the barn, with separate air-spaces, were considered for the measurement (Fig. 2), and were investigated separately for their CH_4 emissions. Each compartment was designed for 48 dairy cows offering a total area available per cow of 10 m^2. Having no solid eave-side walls, the building is naturally cross-ventilated. However, there was a facility to close the western eave-side of the building with curtains. The curtains were open during the summer, partly open in spring and autumn, and closed during winter. The barn had a slatted floor with subfloor storage of liquid manure, and a robot system for fully automated water cleaning of the slatted floor. The two power take-off mixer with electric motors (7.5 KW) for homogenization of the liquid manure beneath the slatted floor were located at the gable wall, next to stable section 1 (Fig. 2). This resulted in a high intensity of homogenization of liquid manure in stable section 1 (‘intensive mixing case’), and a low intensity of homogenization of liquid manure in stable section 2 (‘no intensive mixing case’).

There were 96 lactating Holstein dairy cows in the compartments, with an average milk yield of 34 kg (28-39) per day, and an average live mass of 700 kg (550-870). Cows in the measurement-relevant sections were between the 95^{th} and 190^{th} day of lactation. The cows were fed once a day with a grass and maize silage-based mixed rations, and were able to get concentrate feed at concentrate stations additionally, according to their production (2.5 kg per cow and day on average). The total average dry matter feed intake per cow was 19 kg. The mean crude protein of the mixed ration was 16.6 % (dry matter) and crude fiber was 17.4 % (dry matter).

Figure 2: **Layout of the dairy barn with measuring units** (where D are dosing points for tracer gas injection, S are sampling points, C are concentrate feeders, SB is sampling background and T is temperature measurement)



Source: authors' own illustration.

6.2.2.2 General procedures

Measurements were conducted from December 2010 to December 2011, covering all seasons of the year and various weather conditions.

Gas concentrations were measured in the exhaust air of the compartments. Owing to the large open walls of the barn, the air-outlet location was highly dependent on wind direction. Considering the regional conditions, it was assumed that the exhaust location for measurement of gas concentrations was at the eastern eave-side of the building. Nevertheless, only those time periods (daily basis) when the wind direction led to a west to east cross-ventilation were taken into account, the rest was discarded. In 2011, about 50% of the time period could be used for the analysis. Methane and ammonia emissions from the barn were calculated on average for each season leading to the annual average in equal parts.

6.2.2.3 Measurement of gas concentrations

Measurements of gas concentration were carried out for more than 300 days, recording exhaust concentrations of CO_2 and CH_4 for each compartment. Each compartment was equipped with eight sampling points, in line above the feed alley, put together into one aggregate sample for each compartment. The background (incoming) air was sampled at the western side of the building (Fig. 2). The exhaust air of the compartments and the background air were sampled by vacuum pumps through separate polytetrafluoroethylene (PTFE) sampling tubes into PTFE sample bottles. The sample bottles, the multiplexer (used for switching between samples) and the gas analyzer were placed in the adjacent building (multi-gas analyzer 1412, and a multiplexer 1303 Lumasense Technologies SA, Ballerup, Denmark). On the distance between the barn and the adjacent building the sampling tubes were laid underground and heated. This procedure was performed in order to offer constant measuring conditions throughout the whole year and further to avoid condensation.

The gas analyzer was sent to the manufacturer for calibration after 4 weeks due to a drift in methane concentrations and afterwards every 6 months. In order to check the accuracy of the measurement system in the meantime, calibration gases with known concentrations were used after 4 weeks.

6.2.2.4 Measurement of air volume flow

The air exchange rate was calculated using the tracer decay method (NIEBAUM, 2001; SAMER et al., 2011; SCHNEIDER et al., 2006; SEIPELT, 1999) with a SF_6 electronic capture detector, and converted subsequently into volumetric air flow, per cow per hour. The tracer gas was released as a line source at the windward side of the barn at a height of 4 m from the floor, which allowed proper mixing of the tracer within the compartment (Fig. 2). The sampling system used for the tracer gas measurement was the same as used for the gas concentration measurement. Tracer gas measurements were performed during summer with open curtains when a cross ventilation (west to east) was given. Based on wind direction and wind velocity data, the air exchange rate and the volumetric air flow rate could be estimated for the periods of cross-ventilation. The volumetric air flow rate was determined on an hourly basis considering the average wind velocity per hour.

In the case of closed curtains, the CO_2 mass balance, according to CIGR (2002), was applied to calculate the volumetric air flow.

6.2.2.5 Calculation of emissions

The emission rates E [mg h^{-1} cow^{-1}] were calculated on an hourly basis, with the measured gas concentrations and the calculated volumetric air flow rate Q_m [m^3 h^{-1} cow^{-1}], using the following equation:

$$E = Q_m * (C_{in} - C_{out}).$$

Where C_{in} [mg m^{-3}] is the exhaust concentration and C_{out} [mg m^{-3}] is the background concentration of the relevant gas. Multiplying E by the global warming potential of CH_4 (21) leads to emission quantity in CO_2-equ (UBA, 2009: p.57).

6.2.3 Procedure of comparison

The specific farm characteristics of Haus Riswick were implemented into the model, in order to simulate the identical farm for comparison of results on CH_4 emissions.

Emission factors taken from IPCC (2006) were also elected, corresponding to the average annual temperature of 9.8°C, and an average live-weight of 700 kg per cow. Limited to the system boundaries of the experimental farm installation, only emissions from lactating cows were comparable. Furthermore, only high phase lactating cows, between the 95th and 190th day of lactation, were held in the investigated sections of the barn. Implementing a phenotypic milk yield potential of 9,600 kg per cow per year, results in a model per day lactation parameter of 0.354% of yearly milk yield (34kg/9,600kg=0.354%) for the high lactation phase, which is necessary for feed requirement functions of the herd. For comparison, the daily output parameter derived from HUTH (1995) for high lactation phase is 0.33% of yearly milk yield. Considering that only highly lactating cows were held in the relevant stable sections, a milk output potential per stable place of 12,410 kg/year (34 kg * 365 days) is assumed. Referring to the barn characteristics on Haus Riswick, the model was adapted to only simulate emission amounts from lactating cows, on slatted floors with a full-year subfloor manure storage capacity. The simulation horizon also corresponds to the measurement interval of one year on Haus Riswick.

Farm simulations were done for a farm implementing the above-stated farm characteristics, and using each of the explained GHG indicators separately. This leads to different emission estimates depending on the calculation rules of the specific indicators.

6.3 Results

The results enabled the evaluation of the CH_4 emission calculation accuracy of the different model-defined GHG indicators. Table 2 shows the estimated CH_4 emissions per cow and per kg of milk, respectively. CH_4-measurements of the stable sections, denoted above, with and without intensive mixing of liquid manure, are displayed separately. Furthermore, an average case for manure handling is made by taking the average over both measurement districts.

Table 2: **Per year CO_2-equ. derived by different indicators and results of real measurement on Haus-Riswick.**

model results of different indicators					real measurement		
Unit	actBased	prodBased	genProdBased	Nbased	no intensive mixing	intensive mixing	average
[kg CO_2-equ./cow]	2,898	4,637	4,247	3,600	3,570	4,120	3,845
[kg CO_2-equ./kg milk]	0.234	0.374	0.342	0.290	0.288	0.332	0.310

Source: own calculation and illustration.

As illustrated in Table 2, online measurements for CH_4 release lie between 3,570 kg and 4,120 kg CO_2-equ. per cow per year. Obviously, a high mixing intensity of manure leads to overall CH_4 emissions from the barn, 15.4% higher than in the case of low manure homogenization. Dividing the average CH_4 emissions of 3,845 kg CO_2-equ. by the yearly milk yield potential per stable-place of 12,410 kg leads to 0.310 kg CO_2-equ. per kg of milk on average for the experimental installations on Haus Riswick. Accordant calculation results by the model show partial great differences, accounting for 2,898 kg up to 4,637 kg CO_2-equ. per cow for the identical farm. Estimates by the *actBased* indicator lie below the measurement values. The results from Table 2 are taken to quantify the absolute and relative deviations of indicator GHGs from the actual measured CH_4 emissions. Measurements from the stable part with and without intensive mixing of manure are taken as a representation of lower and upper boundaries of actually occurring emissions, depending on the intensity of manure homogenization.

The comparison of indicator-derived CH_4 emissions with measurement results is shown in Table 3. Compared with ‘no intensive mixing’ measurements, the NBased indicator leads to the most adequate CH_4 estimates, with only a slight overestimation of 0.9%. As the defined upper bound by the ‘intensive mixing’ stable section, with 4,120 kg

CO_2-equ. per cow, is 15.4% higher than the lower bound, the overestimation of the indicators prodBased and genProdBased diminishes. The NBased estimation is even 12.6% below the measured upper value. In contrast, the underestimation of the actBased calculation increases to 29.7%, when compared with the measurements from the intensively homogenized stable section.

Table 3: **Deviations of indicator results from real measurements**

		actBased	prodBased	genProdBased	Nbased
no intensive mixing					
absolute deviation per cow	[kg CO_2-equ.]	-672	1,067	677	31
absolute deviation per kg of milk	[kg CO_2-equ.]	-0.05	0.09	0.05	0.00
relative deviation	%	-18.8%	29.9%	19.0%	0.9%
intensive mixing					
absolute deviation per cow	[kg CO_2-equ.]	-1,222	517	128	-519
absolute deviation per kg of milk	[kg CO_2-equ.]	-0.10	0.04	0.01	-0.04
relative deviation	%	-29.7%	12.6%	3.1%	-12.6%
average mixing intensity					
absolute deviation per cow	[kg CO_2-equ.]	-947	792	403	-244
absolute deviation per kg of milk	[kg CO_2-equ.]	-0.08	0.06	0.03	-0.02
relative deviation	%	-24.6%	20.6%	10.5%	-6.4%

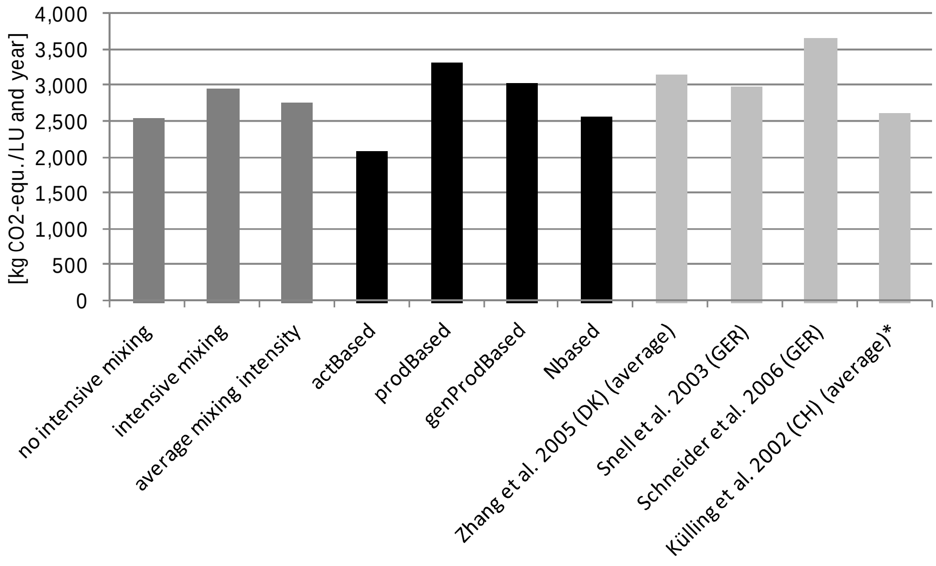
Source: own calculation and illustration.

The estimates from the actBased indicator lead to a clear underestimation of actual emissions per cow, occurring from the stable section with low manure homogenization. The model-calculations by the prodBased and genProdBased indicators even overestimate the upper bound. However, overestimating the online-measurements by only 3.1%, the genProdBased indicator can be identified as a good proxy for dairy cow emissions, with high rates of subfloor manure homogenization for our specific farm.

Comparing the average of the measurements from both stable sections with the model results, the prodBased estimator routinely overestimates the real emissions by large amounts (20.6% on average). The NBased indicator scheme underestimates the average CH_4 values by about 6.4%, whereas the actBased one leads to an aberration of -24.6%. The actBased indicator, routinely, has negative deviations, while the prodBased and genProdBased indicator schemes have positive deviations from actual measurements. Only the calculations of the Nbased indicator lie between the upper- and lower-bound of actual measurements. Considering these results, the genProdBased indicator seems to be an adequate proxy for the upper bound of the measured emissions from the stable, with high homogenization intensity of liquid manure. The NBased indicator shows the highest accuracy in CH_4 calculations for the lower bound, defined by the barn section with low movements of manure, and even emerges as a good proxy for the average emissions per cow measured over both stable sections (average).

As seen in Figure 3, indicator estimates of the NBased lie between the minimum and maximum of measurements from Haus Riswick. Furthermore, model estimates can be compared with findings from the literature, bearing in mind the limited usability of literature findings as stated beforehand. Therefore, model estimates, as well as online-measurement results from Haus Riswick, are expressed as emission amounts per LU, comparable with findings reported in the literature. Emission inventories per LU derived from the literature are higher compared to long-term measurements from Haus Riswick. Only the estimates from KÜLLING et al. (2002) are comparable to the measured amounts. This underlines the gain in validation accuracy of the model approach of DAIRYDYN, by using one-year online-measurements instead of literature information, as mentioned in the introduction.

Figure 3: **Visualization of model results compared to real measurements and literature findings for slatted floor conditions** (measured and simulated emissions are rebased to emissions per LU (500 kg of live weight); DK: Denmark, GER: Germany, CH: Switzerland, *average is built over all investigated feeding strategies and according measurements with 14-day manure storage time)



Source: own illustration.

Model estimates by the actBased indicator lead to the lowest emission quantities, whereas the prodBased indicator scheme results in comparatively high estimates. By modeling the identical farm as presented by the experimental barn on Haus Riswick, the NBased indicator leads to CH_4 calculations near the 'real world' quantities.

6.4 Discussion

The results show that the range of model estimates for CH_4 emissions for the dairy barn on Haus Riswick is quite broad, varying between 18.8% below (actBased) the lower bound to 12.6% above (prodBased) the upper bound.

The overestimation of CH_4 by the prodBased indicator is a result of the construction of the indicator-specific emission parameter per kg of milk. The per kg emission parameter was derived by dividing the IPCC Tier 1 default value per cow by a potential milk yield of 6,000 kg per year. This routinely overestimates real emissions by multiplying the emission parameter by the actual milk yield level of 12,410 kg per stable place on Haus Riswick, assuming a constant per kg milk emission factor with increasing milk yield. Hence, approximation of CH_4 emissions, using the prodBased indicator, are quite inconsistent if dairy facilities are modeled that deviate from a 6,000 kg average milk yield potential per cow. Following the results in Table 3, using the actBased indicator (meaning default Tier 1 IPCC CH_4 parameters per animal) leads to underestimations for a farm with high milk yield potential, owing to the default emission parameters appropriate for a 6,000 kg milk yield potential. As further shown in Table 3, the genProdBased indicator derives good estimates in the case of the stable section with high manure homogenization rates. On average, the model CH_4 calculations, using the NBased indicator, produce the best proxy for actual measured CH_4 amounts, owing to recognition of higher manure removal frequencies, and adjustment to the real average feed digestibility. Not only the small underestimation of real emissions (-6.4% on average), but also the fact that its estimates lie between the measured upper- and lower-bound for high and low mixing intensity underlines the suitability of the most detailed indicator for CH_4 emission calculation in dairy barns. With regard to Figure 3, the NBased estimates also lead to per LU emissions comparable to results from KÜLLING et al. (2002) (only -1.3% deviation), which further underlines its accuracy and adaptability to other farm types, because experimental attributes of KÜLLING et al. are comparable to the specified model experiments (KÜLLING et al. investigated high lactating cows with a lactation of about 31.3±5.1 kg milk d^{-1} and an average live weight of 635±56 kg cow^{-1}).

As the actBased indicator falls back on the most aggregated process variables, and represents a default and very simple emission accounting, the emission approximation increases in accuracy compared to real measurements, when incorporating more detailed process variables into the indicator scheme.

This result is in line with findings from ELLIS et al. (2010), who compare GHG simulation equations with small-scale measurements on dairy cows. They state that the simple Tier 1 approach of IPCC, equivalent to our actBased indicator, leads to the worst emission estimates in contrast to the NBased one (comparable to Tier 2 methodology of IPCC), which was also valued as relatively adequate by these authors (p. 3251). Nevertheless, estimated errors are still rather high, but more detailed approaches have been missing up to now. (ELLIS et al., 2010: p.3250) As about 80% of the dairy barn CH_4 emissions stem from animal rumination it is obvious that indicators with detailed accounting of feeding patterns and milk output intensity (NBased) lead to more accurate CH_4 calculations (ELLIS et al., 2010). This divergence in GHG accounting accuracy between default and detailed indicators even increases the stronger farm characteristics deviate from attributes the simple default emission factors (actBased/Tier 1) are calibrated on.

The comparison of indicator-modeled CH_4 emissions with online measurements should lead to a validation of the DAIRYDYN model. Compared with findings of other studies, the model results — except when using the actBased and prodBased indicators — offer relatively moderate deviations (between -6.4 and 10.5%) from average actual CH_4 amounts. For example TALLEC and HENSEN (2011: p.6) underestimate real CH_4 emissions by about 25%.

However, it should be noted that the actual measurement results of Haus Riswick may also include minor measurement errors. For example, the CO_2 mass balance method for the estimation of the air exchange rate bears the risk of inaccuracy, since — beside the cows — there may be other minor CO_2 sources within the barn (e.g. manure, feed and/or machines). Furthermore, it has to be considered that Haus Riswick represents a well-managed demonstration farm, having very well-balanced feed rations and performing high-frequency cleaning of surfaces within the barn. It can be assumed that, in practice, not all farms are able to fulfill best agricultural practices, and that they may have slightly higher emissions. Unfortunately, up to this point, we were not able to quantify the portion of difference between measured and calculated CH_4 occurring from the modeling bias or the measurement error.

6.5 Conclusion

Concluding from the former sections, this study underlines that generally the CH_4 calculation schemes implemented into the model DAIRYDYN lead to good approximations of actual stable CH_4 release. The highest accuracy in CH_4 approximation for the experimental farm is given by the most detailed indicator (NBased).

Although the different indicator schemes within the model approach of DAIRYDYN may show adequacy in emission accounting to some degree, the usefulness for political GHG control instruments is not yet given. The validation of the model, using different GHG indicators in this study, is only representative of one specific lactation level and stable type. Hence, further research has to be done to compare modeling results for other intensity levels and stables. Therefore, our study underlines the advantage of using long-term measurements of a whole stable system for a high number of animals to ensure representative estimates including variability within the cow population and the influence of exogenous parameters over time (e.g. feed quality, temperature…). Special emphasis should therefore be placed on the use of long-term measurements for model validation instead of using small scale and short term results.

Furthermore, it has to be emphasized that management options are a relevant variable to include into modeling approaches. Limiting calculations to default and highly aggregated GHG calculation schemes may be inadequate for a broad range of dairy farm types due to the high heterogeneity in the actual farm population.

Certainly, adequate emission accounting is of great relevance (ELLIS et al., 2010: p.3246). However, in the case of the enforcement and control of emission ceilings in agricultural dairy production, induced abatement strategies by the different indicators are of great interest, leading to different cost implications for the abatement of GHG amounts. Hence, further research has to be done in this field, capturing engineering costs at the farm level, as well as administrative costs for control and enforcement.

Also, the model approach has to be developed further to increase the level of detail (e.g. as done by BANNINK et al. (2011), implementing a more detailed IPCC Tier 3 approach for dairy cow CH_4 estimation). This is of special interest not only for ruminant CH_4 emissions but also for the emissions occurring from manure, as the diet composition also significantly impacts the CH_4 amount stemming from the animals' excreta (HINDRICHSEN et al., 2005; KÜLLING et al., 2002).

The inclusion of more detailed information from the production process, in order to obtain less biased emission estimates, hence guarantees more reliable results for a more diversified range of dairy farms, especially if willing to use modeling results for more aggregated and political purposes.

In general, our study showed that the exchange between and the combination of modeling and measuring science is a valuable cooperation, offering the possibilities to improve the accuracy in modeling and to amend or partly replace the time and cost intensive measurements in the future.

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Chapter 7: **Comparison of GHG-emission indicators for dairy farms with respect to induced abatement costs, accuracy and feasibility**[41]

Abstract

There is broad debate about including agriculture in greenhouse gas (GHG) reduction efforts such as the European Emissions Trading Scheme. Since most agricultural GHG emissions originate from non-point sources, they cannot be directly measured, and therefore have to be derived by calculation schemes (indicators). We designed five such GHG indicators for dairy farms and analyzed the trade-offs between their feasibility, measurement accuracy, and level of induced abatement costs. Analyses of induced abatement costs and calculation accuracy are based on emission reduction simulations with a highly-detailed single-farm optimization model. Feasibility is discussed in a qualitative manner. Our results indicate that the trade-offs depend on both farm characteristics and the targeted reduction level. In particular, the advantages of detailed indicators decrease for higher abatement levels. Only the least feasible indicator led to abatement costs that would result in emission efforts at given prices in the European Emissions Trading Scheme, although with a rather small potential. Our results thus suggest little potential for including dairy production into market-based reduction policies.

Keywords: *emission indicators, GHG accounting, farm level measurement, capability of indicators.*

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[41] This chapter is a pre-copy-editing, author-produced version of an article accepted for publication in *Applied Economic Perspectives and Policy* following peer review. The definitive publisher-authenticated version "LENGERS, B., BRITZ, W. and K. HOLM-MÜLLER (2013): Comparison of GHG-emission indicators for dairy farms with respect to induced abatement costs, accuracy and feasibility, Applied Economic Perspectives and Policy 35(3):451-475" is available at: http://aepp.oxfordjournals.org/content/35/3/451. The research was funded by a grant from the German Science Foundation (DFG) with the reference number HO 3780/2-1. Special thanks also to three anonymous reviewers as well as the editor of the AEPP journal for helpful suggestions and a straight forward reviewing process.

7.1 Introduction

With the adoption of the Kyoto Protocol in 1997, 37 industrial nations and the European Community agreed to reduce greenhouse gas (GHG) emissions before 2012 by about 5% relative to GHG levels in 1990, including emissions from the agricultural sector (UNFCCC, 2009; UNFCCC, 2013). New reduction goals were to be enacted during the 2012 UN climate conference in Qatar. To pursue these goals, agriculture, which emits an estimated 10-12% of yearly global GHGs (NIGGLI et al., 2009: p.1), will probably be required to contribute to national and global emission reduction aims. Policy instruments such as tradable emission permits, emission taxes or statutory requirements, which have already been implemented in industrial sectors, might thus be expanded to the agricultural sector. New Zealand already plans to indirectly incorporate agricultural emissions from livestock into its Emissions Trading Scheme (ETS) (MPI, 2013). Furthermore, the Kyoto Protocol’s international guidelines also include agricultural GHGs in their reporting mechanism (UNFCCC, 2008).

The design of policy instruments targeting GHG abatement in agriculture faces a number of combined challenges. First, in contrast to industrial sectors, agriculture is characterized by many small firms with non-point diffuse gaseous emissions of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2), where a direct measurement is not practical (OSTERBURG, 2004: p.209). Therefore, emissions must be accessed via accounting schemes (indicators) drawing on observable attributes of the investigation unit (e.g., animal number, crop acreages, fertilizer use, fodder ingredients, etc.). This is already an established approach for controlling other non-point externalities from agriculture, specifically in the context of the EU nitrate and water directives (EC, 1991; EC, 2000).

Second, policy will have to decide about whether, how much and how to include agriculture in abatement efforts. Knowledge about abatement, monitoring and control costs (curves) per unit of GHG removed in comparison to other sectors’ costs (curves) is required to determine the extent of agriculture’s participation. In the EU, ETS permits have been traded at around 7 €/ton in 2012 (VAN RENESSEN, 2012), reflecting current marginal abatement costs (MACs) in the included sectors. As we will show for the example of dairy farms in this paper, the abatement level also impacts differences in accuracy and abatement costs (ACs) between indicators. Therefore, agriculture’s share in abating GHG emissions will also determine the performance of various indicators.

A related question is the choice of the appropriate policy instrument. Theoretically, market-based instruments such as emissions trading are made cost efficient by endogenously equating abatement costs between different emitters. But in agriculture, these instruments require an indicator which might be costly to monitor and control. Statutory requirements that render low-emission farming practices mandatory, for example, are less flexible but might not require an indicator and are typically easier to monitor and control. Against this background, we aim to generate important information for rational policy design by quantitatively investigating the impact of abatement level, farm attributes, and indicator construction on abatement costs, measurement accuracy, and data feasibility (WALZ et al., 1995; OSTERBURG, 2004: p.214).

In contrast to our paper, most of the existing studies on accounting and mitigation of GHGs from agriculture provide results for one specific indicator, in most cases based on IPCC (Intergovernmental Panel on Climate Change) formulas or default values (e.g., DECARA et al., 2005; LESSCHEN et al., 2011; OLESEN et al., 2006: p.209).

These studies confronted policy-makers with abatement cost curves derived with different methodologies such as engineering approaches, supply side models, and market models, and based on different indicators, all of which yielded quite different MACs. KESICKI and STRACHAN (2011) thus conclude that not enough attention has been paid to the impact of methodological and further choices on MACs curves. SCHNEIDER and MCCARL (2006: p.285) also stress that the underlying methodological assumptions should be examined carefully when economic results for GHG reductions are interpreted. The relations between abatement costs and indicators especially call for a deeper investigation.

Against this background, LENGERS and BRITZ (2012) highlighted the fact that GHG indicators influence chosen abatement strategies and adherent costs; their publication focuses on the description of DAIRYDYN (dairy dynamic), which is a bio-dynamic model approach for dairy farms, and offers some illustrative applications that show how emission reduction strategies and their related profit losses depend to a large extent on the GHG indicator chosen. We will now use the same model to investigate the performance of different indicators systematically by varying abatement levels and characteristics of the farms considered. This will allow us to contribute to a better understanding of the interplay between abatement level, indicator accuracy, induced abatement measures and costs, and farm characteristics for the different indicators. We will also explore the differences between abatement costs that are relevant to society (these are related to actual emission reductions, and called societal abatement costs), the abatement costs that are relevant at the farm level (these are related to emission reductions calculated via an indicator, and called on-farm abatement costs), and their relation to the factors mentioned above. If farms are rewarded according to indicators with a measurement bias, the performance of economic instruments like taxes or emission trading could be seriously affected and lead to adverse consequences.

The main aims of the paper are thus to quantitatively examine trade-offs between accuracy, level of induced abatement costs, and the feasibility of monitoring and controlling the required data for the different GHG indicators to collect information on indicator construction and its specific usability in a political context. Empirical data about monitoring costs and indicator feasibility are currently nonexistent, meaning that these points can only be qualitatively discussed here.[42]

This paper is organized as follows: the next section offers a review of existing studies that apply specific GHG accounting schemes, from which the requirements of suitable indicators are derived. Next, we describe the bio-economic model DAIRYDYN and its application in our context. Using variously detailed farm-level data we define different indicator schemes drawing on IPCC (2006) guidelines, which we then relate to abatement options covered in the model. The results section highlights differences in GHG abatement costs and GHG estimation accuracy between the indicators for a systematic variation of key farm attributes and abatement levels. This information leads to a brief discussion on the practical applicability of these indicators from the viewpoint of policy implementation, as well as the perspective of farmers facing GHG policy instruments. Finally, we will summarize and draw policy conclusions from the obtained results.

7.2 Literature review

Indicators for the quantification of not directly measurable environmental externalities have been used extensively in different industry sectors (AZAPAGIC, 2004; IPCC, 2006; NIEMEIJER and DEGROOT, 2008; RIDGLEY, 1996), not least in the context of policies related to mitigating GHGs. A vivid discussion is now taking place regarding whether and how agriculture could be included in GHG abatement efforts; if market-based policy

[42] To our knowledge, only ANVEC (2011) tried to estimate transaction costs for GHG control in agriculture, however by drawing on observed transaction costs for industrial sectors with large scale, point source emitters. Given the differences in what processes need to be monitored in industrial sectors and on a farm to estimate GHGs, Anvec's approach provides solely a minimum level and does not discern between different indicators.

instruments such as emission trading or taxes are used, an indicator system acting as a technological control parameter for GHG emissions at the farm level is required (OSTERBURG, 2004: pp.211; SCHEELE et al., 1993: p.298). Research work regarding emission inventories and potential GHG abatement in agriculture uses indicators to calculate emissions from agricultural activities, and in some cases also derives mitigation-related MACs (e.g., BREEN, 2008; DECARA et al., 2005; HEDIGER, 2006; OLESEN and SCHELDE, 2008; PÉREZ and BRITZ, 2003; SCHILS et al., 2005). CROSSON et al. (2011) presented an overview of 31 published studies of GHG emissions from dairy and beef-producing farms, and emission calculations in the reviewed studies were overwhelmingly based on IPCC equations or default values (IPCC, 2006 or earlier versions). Only a few of the studies developed their estimations from experimental emission measurements on farms (e.g., JUNGBLUTH et al., 2001; NGWABIE et al., 2009).

The level of detail regarding GHG emission calculations in the different studies varies greatly, depending on the availability of data and the research goal. BREEN (2008) based his calculations exclusively on animal numbers. Similarly, MACLEOD et al. (2010) used fixed emission factors per unit of livestock or area of land. VERGÉ et al. (2007: p.683) quantified the 2001 GHG emissions of the Canadian dairy sector in two ways: per animal, and as a function of milk yield. In a study estimating GHG emissions from agriculture for the German Federal State of Baden-Württemberg, NEUFELD et al. (2006: p.239) found that, with an R^2 of 0.85 and $p<0.01$, the stocking rate seemed to be a sufficient indicator if activity units (animal herds and total fodder acreage) reflect “true” values.

CLEMENS and AHLGRIMM (2001) used emission equations for CH_4 from ruminants regressed by KIRCHGESSNER et al. (1993) based on raw nutrient intake, as well as milk yield, body weight, and type of roughage (KIRCHGESSNER et al., 1995) to discuss reduction potentials of abatement options in animal husbandry. Conclusions about N_2O released from excreta are drawn following N-excretion functions from KIRCHGESSNER et al. (1993) based on milk yield potential and crude protein content of the forage.

DECARA and JAYET (2000) assessed greenhouse gases and possible abatement costs for the French agricultural sector using rather simple equations from SAUVANT et al. (1996) based on the gross energy intake of feed to calculate methane emissions from ruminants. Additionally, DECARA and JAYET (2000) used an equation from BOUWMAN (1989) for N_2O quantification, which is solely based on total N fertilizer application.

Thus, available studies have used quite different indicators regarding the level of detail and the aggregation of relevant input variables. However, as each study uses just one indicator, they are unable to analyze how various designs in accounting methods impact their emission estimates and abatement costs if GHG reductions are implemented.

To the best of our knowledge, DURANDEAU et al. (2010) were the first to examine the influence of different detailed emission accounting schemes on MACs. However, they did this solely for N_2O from synthetic fertilization, and concluded that for an 8% emission reduction, induced MACs with a second-best indicator were about 7 times higher than with a first-best, more detailed emission scheme. The impact that the GHG indicator construction has on the possibilities of low-cost mitigation on the farm level was also shown by LENGERS and BRITZ (2012), based on illustrative simulation results of GHG abatement costs on dairy farms. Besides the differences in induced *net on-farm*[43] mitigation costs between indicator schemes, LENGERS and BRITZ (2012) also pointed out that the measurement accuracy directly impacts its difference to the *net societal*[44] mitigation costs.

To conclude, different sets of indicators can be found in the literature, but to date no systematic comparison exists of different possible indicators concerning important criteria.

Criteria for appropriate indicators can be derived from a number of sources, for example BACH et al. (2008: p.10), DÖHLER et al. (2002: p.30), EUC (2001: p.10), HALBERG et al. (2005), HOLM-MÜLLER and ZIMMERMANN (2002), KRISTENSEN et al. (2009: pp.15-16), OECD (1999: p.19), OSTERBURG (2004: pp.210-211) and WALZ (2000). The most important requirements can be summarized using three criteria: *feasibility, accuracy,* and *low-cost abatement. Feasibility* refers to the data requirement at the farm level for monitoring and level of control. Hence, feasibility depends on the existence or potential of developing farm-level reporting systems for an indicator, and generally diminishes with increasing data requirements. *Accuracy* is linked to the ability of emission indicators to approximate actual emissions, and thus relates to the detail and consistency of calculation schemes (SCHRÖDER et al., 2004: p.20). *Low-cost abatement* is defined as the ability of an indicator to trigger an abatement strategy for a specific reduction amount that provokes lower abatement costs than other indicators. In our study this point is discussed from an on-farm and societal perspective, including: (1) net on-farm costs, that is, the on-farm abatement costs that depend on the expenditure and mitigation potential of the abatement measures covered by an indicator (related to calculated GHG emissions as

[43] Notation comparable to the not-normalised abatement costs in the study by LENGERS and BRITZ (2012).

[44] Notation is equivalent to the normalised abatement costs by LENGERS and BRITZ (2012).

opposed to actual); and (2) net societal costs, that is, the on-farm abatement costs related to actual GHG abatement. Hence, these costs depend both on the abatement options covered by an indicator and a possible difference between accounted and actually emitted GHGs, and therefore on indicator accuracy. Trade-offs between these three requirements typically require compromises in the design of an adequate GHG indicator scheme (WALZ et al., 1995).

7.3 Methodology

Overview

Our aim in this paper is to compare a set of indicators estimating GHGs from dairy farms by quantifying their ACs, MACs, and their accuracy with regard to emission estimates at different abatement levels. In order to do so, the following step-wise approach is chosen. First, a sufficiently detailed single profit-maximizing farm model is represented (see also appendix 1). Second, variously detailed GHG indicators are developed, each with their specific set of credited abatement options, and then integrated in the simulation model. Next, for each indicator the model is confronted with increasing abatement levels to quantify MACs and ACs. Finally, a set of experiments over a range of two central farm attributes—herd size and milk yields—deliver results on specific MACs and ACs based on maximum profit level changes in farm plans. The simulated GHG reductions and MACs allow us to analyze abatement cost and accuracy aspects of the indicators. The feasibility of indicators is not analyzed in a formal manner, but rather briefly discussed.

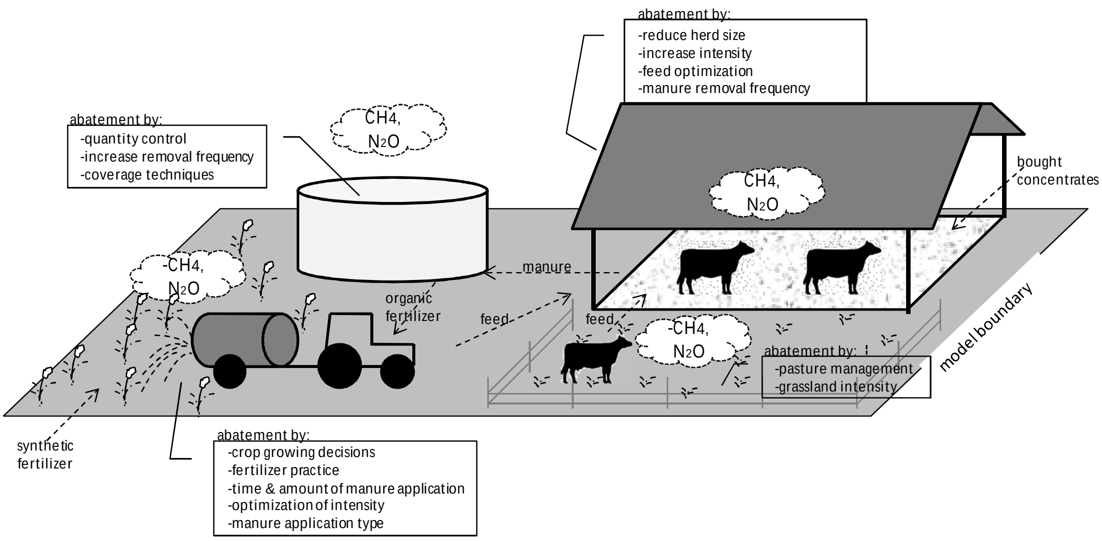
7.3.1 The single farm optimization model

The DAIRYDYN model (LENGERS and BRITZ, 2012) applied in this paper is a fully dynamic mixed integer linear programming model of a single dairy farm that maximizes the expected net present value of future profits over a sequence of years and different states of nature. This model covers decision variables that impact GHG emissions, including variables such as herd sizes and milk yields, crop acreage and yields, manure handling, fertilizer use, and feeding practices (see figure 1), where appropriate also with a monthly resolution. Different GHG emission indicators (see next section) attach their specific emission factors to these decision variables and can be used to constrain the model by an indicator-specific GHG emission ceiling. This in turn allows ACs to be derived for each indicator (VERMONT and DECARA, 2010). As the decision-maker of each farm is assumed to be fully informed and rational, simulation results entail best-practice behavior. The mixed integer approach reflects indivisibilities in investment and labor-use decisions, whereas the fully dynamic character accounts for path-dependencies in production and investment decisions. Compared to so-called engineering models that evaluate abatement options independent of each other, the programming approach endogenously reflects interactions between different GHG mitigation strategies with regard to the farm program, and thus reflects changes in profits as well as GHGs emitted. Examined bio-physical interactions, based on detailed response and mass flow functions, include possible trade-offs between emissions of different gases from identical or different sources (WEYANT et al., 2006), which are important for consistency (MACLEOD et al., 2010: p.200). A simulated abatement strategy thus consists of a maximum profit mix of different abatement options and their respective levels (e.g., amount of GHG-reducing feed additives per cow). The costs of such a mix can markedly differ from the independent appraisal of single abatement options, as is shown by SCHNEIDER and MCCARL (2006) using the Agricultural Sector and Greenhouse Gas Mitigation Model (ASMGHG).

In contrast to partial equilibrium models (e.g., PÉREZ, 2005), which typically work on a more aggregated scale, in this farm model price changes are exogenous.

The system boundary (figure 1) of the model is the farm gate, such that all GHGs emitted by farm activities (animal husbandry, crop production, manure handling, etc.) are covered. However, emissions linked to the production of purchased inputs such as fertilizer, diesel or concentrates, or to the processing and marketing of outputs are not taken into account. The approach is hence not a life-cycle assessment, but is rather in line with the accounting logic of the GHG inventories under the Kyoto Protocol (IPCC, 2006).

Figure 1: **Boundary of the system approach**



Source: own illustration.

7.3.2 GHG indicators

All implemented GHG indicators are based on IPCC (2006) guidelines that underlie the reporting obligations of parties to the Kyoto Protocol. These indicators can be assumed to be scientifically accepted and consistent (e.g., by avoiding double-counting). The guidelines offer fundamental emission parameters and calculation schemes for different detailed accounting systems: from the simple Tier 1, to the most detailed Tier 3. Our indicators reflect German conditions in accordance with these guidelines.[45] However, we use deviating background emission factors for agricultural soils[46] for N_2O (VELTHOF and OENEMA, 1997: p.351) and for CH_4 (BOECKX and VAN CLEEMPUT, 2001).

Based on this IPCC tier approach, we develop various detailed indicator schemes, all of which reflect whole farm emissions based on GHGs emitted from enteric fermentation, manure management, soil cultivation (background emissions for arable and grassland), and fertilizer use (see appendix 2). Here, we only briefly explain differences between the indicators (see appendix 2 for more details, and LENGERS (2012a) for full documentation); these differences determine data demands and how they relate farm attributes and decision variables to the accounted GHG emissions. Compared to LENGERS and BRITZ (2012), we use a highly detailed calculation scheme as the reference indicator.

The simplest GHG indicator is named *actBased*, and is equivalent to the indicator used in Tier 1 methodology; single-default emission factors (CH_4, N_2O) per activity unit in crop or livestock production are multiplied by the activity levels (e.g., number of cows, ha maize silage).

The second indicator (*prodBased*) is derived from the activity-based indicator, but differs in the calculation of emissions from lactating cows and crop production. Here, the Tier 1 emission parameter per cow is divided by an average milk yield to derive a default per kg milk emission parameter. Similarly, the default per ha emission factors for crops are divided by average crop yields to arrive at the default per unit of output emission factors. Thus, GHGs depend on output quantities with emissions per cow or ha linearly increasing in yield levels.

[45] Up to now, CO_2 emissions are not accounted for because options like land use change, afforestation and change of tillage practices are not implemented in the model approach.

[46] Soil background emission factors for N_2O from IPCC (2006) are based on a study on peat soils with a high organic matter, which does not fit average soil conditions in Germany. The CH_4 background emissions from soils are not recognised by IPCC methodology.

Additionally, the *genProdBased* indicator scheme shows that emissions per kg of milk decrease with increasing milk yield, according to Tier 3 methodology for emissions from enteric fermentation of lactating cows. Based on a detailed estimate of the animal's gross energy (GE) demand, only emissions linked to the cow's energy demand for lactation increase with increasing milk yields, while those linked to maintenance, growth, and activity are distributed over the resulting higher output quantity.

For the calculation of emission parameters used for the prodBased and genProdBased indicator, fix shares of storage types for manure and fixed application shares and types for synthetic and organic fertilizer are assumed.

The *NBased* indicator offers a more detailed calculation scheme. Emissions from enteric fermentation are now derived for all types of cattle by using GE demand requirement functions assuming default feed digestibility. For manure storage, methane and nitrous oxide emissions are calculated based on monthly storage amounts with storage type-specific emission factors (subfloor, surface storage with or without coverage, etc.). The emissions from crop production are based on actual monthly nitrogen applications, differentiated by synthetic and organic fertilizer, the latter also depending on application technique (broad spread, drag hose, or injection).

The most detailed emission indicator is presented by the so-called *refInd* scheme. This *reference indicator* serves as a precision benchmark for the other indicators by building on the *NBased* indicator. However, *refInd* reflects how differences in energy digestibility of the feed ration impact GHG emissions of enteric fermentation (BENCHAAR and GREATHEAD, 2011; HELLEBRAND and MUNACK, 1995; MACHMÜLLER and KREUZER, 1999), and accounts for digestibility improvements from adding fats and oils to the ration.

Out of these specific indicator calculation procedures, various effects arise concerning the decision-makers' affinity to adopt mitigation strategies.

7.3.3 Abatement options recognized by indicators

As shown in table 1, the abovementioned indicators credit only specific abatement options. Thus, farmers will only realize those abatement options which are included in the GHG inventory calculation of the applied GHG indicator, even if others would potentially abate the same amount of emissions at lower costs. The considered options are taken from FLACHOWSKY and BRADE (2007), OENEMA et al. (2001), and OSTERBURG et al. (2009), excluding those banned by German or European law or not supported by scientific findings (see LENGERS (2012b) for more detail). As illustrated by table 1, some options have a more investment-based character (permanent options), whereas others may be changed flexibly during periods or months (variable options).

Table 1: **Indicator dependent choice of abatement options** (indicator relevant options are flagged with an x in the corresponding cells)

	actBased	prodBased	genProdBased	Nbased	refInd
permanent					
manure management techniques				x	x
application techniques				x	x
variable					
fodder optimization				x	x*
breeding activities			x	x	x
intensity management			x	x	x
N-reduced feeding				x	x
fertiliser practice				x	x
area cultivated	x	x	x	x	x
herd size managment, crop growing decisions	x	x	x	x	x
feed additives/ fat content					x
pasture management/ increase grazing		x	x	x	x

(* also recognising digestibility of different feed components)

Source: own illustration.

Generally, farms can adjust to the introduction of an emission ceiling not only by changing decision variables linked to animal production, but also by adjusting arable crop and grassland management.

By definition, our reference indicator covers all GHG mitigation options implemented in the model, thereby allowing for the most flexible and thus inexpensive abatement strategy. At the other extreme, the *actBased* indicator credits only reductions in herd size and/or crop hectares such that the abatement costs are equal to the full gross margins of the activities that were given up. Generally, with growing detail, indicators reflect more abatement strategies and thus open more possibilities for cost-saving reactions to GHG ceilings. Less-detailed indicators drawing on aggregate farm attributes, such as the herd size, offer rather limited abatement strategies—at the extreme only a single one—which could provoke high abatement costs (PAUSTIAN et al., 1997: p.230); this point is also raised by SCHRÖDER et al. (2004: p.20) and SMITH et al. (2007: p.22).

7.3.4 Derivation of ACs

In order to derive ACs, permitted total farm emissions are stepwise reduced compared to the GHGs emitted without restriction, the so-called baseline. The total "net on-farm" ACs are equal to the simulated profit loss against the baseline resulting from the implementation of an emission ceiling. This loss is net of the measurement, administration, and control costs for quantifying emissions. Dividing that profit loss by the emission reduction delivers the average net on-farm ACs per kg CO_2-equ. By stepwise enforcing the ceiling, points on the ACs curves can be simulated. Dividing the change of total ACs between reduction steps by the change in abated GHGs leads to net on-farm MACs per unit of CO_2-equ. The resulting MACs curves, simulated here in 2% steps up to a 40% reduction in GHG emissions, differ between indicators for the very same farm.

As mentioned, there is a difference between net on-farm ACs/MACs and net societal (actual) ACs/MACs because the GHG indicator might account for more or less GHGs than actually emitted. The net societal ACs/MACs are calculated by dividing the loss of profits induced by the specific applied indicator scheme by the "actual" emission reductions. Here, "actual" emission reductions are quantified by using the most accurate accounting scheme, which is the reference indicator.[47] Normalization thus renders the indicators comparable by correcting differences in GHG estimates for an identical production plan.

7.3.5 Model runs

A single simulation run with DAIRYDYN is based on the definition of basic farm characteristics such as the initial herd size, the cows' milk yield potential, age of stables, as well as prices and costs. Farm endowments of land, labor, and machinery are derived from the initial herd size based on engineering rules. Biological reproduction rates and prices both stem from data collections such as KTBL (2010).

We analyzed differences between the indicators for 70 different dairy farms that were formed by varying the herd size between 30 and 120 cows in 10-cow steps, and varying the milk yield from 4,000 to 10,000 kg $head^{-1}$ $year^{-1}$ in 1,000 kg steps. Each farm is simulated over 15 years to potentially cover investment-based mitigation strategies.[48] However, to keep the analysis simple, no re-investments in stables over the planning

[47] For details concerning the normalization procedure, see LENGERS and BRITZ (2012) or appendix 1.

[48] With regard to the abatement costs of measures, the longer time horizon is important to render investment-based and long-term abatement decisions attractive (DEL RIO, 2008).

horizon are needed. We also assumed that neither buildings nor machinery could be sold during the simulation horizon. Otherwise, introducing GHG ceilings might trigger complete farm exit decisions. Such an exit allows GHG emissions to drop to zero at a certain GHG emission target, which renders the calculation of ACs beyond that point useless. Hence, the smooth reaction of the model partly reflects the fact that farmers possess capital endowments that they cannot sell. Additionally, we assumed that farm labor is able to work hourly off farm, albeit at quite low wages.

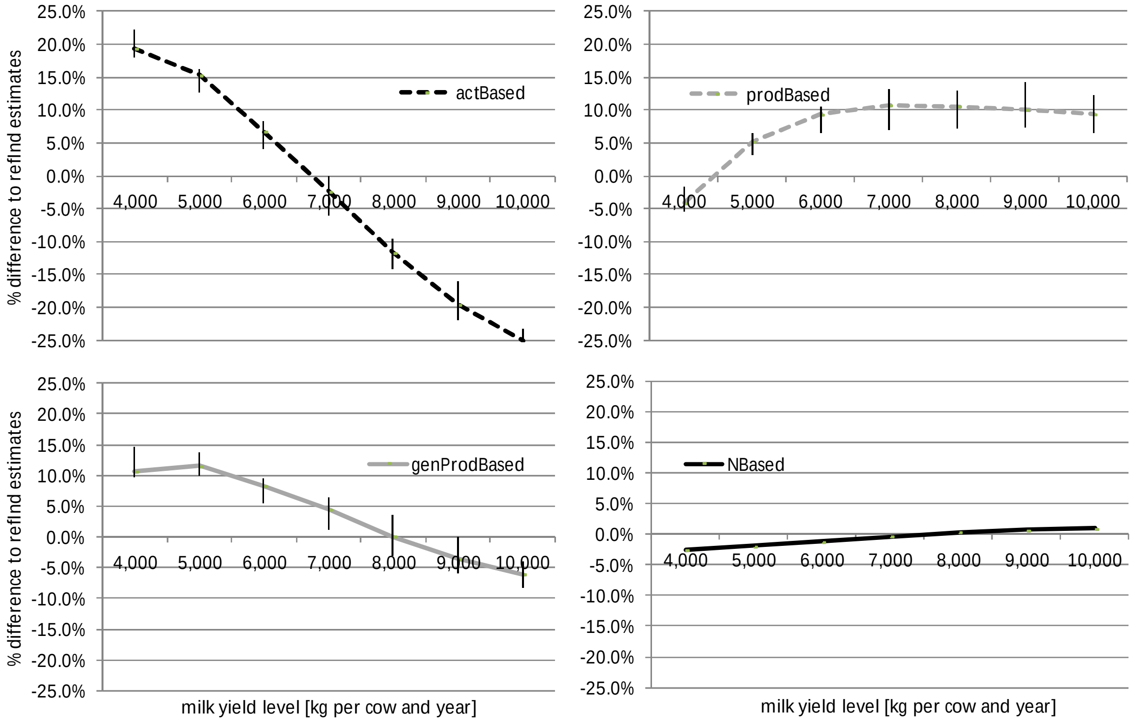
7.4 Results: Evaluation of the indicators

The abovementioned indicators are evaluated below according to the requirements identified in the literature section: their accuracy, potential for low-cost abatement, and their feasibility. We know that when moving to more detailed indicators, accuracy increases and feasibility decreases, while abatement costs should drop. But these relations must be quantified for a decision on the targeted abatement level, the policy instrument, and possibly an indicator.

7.4.1 Accuracy of emission accounting

GHG emissions determined at the farm level will typically differ for the very same farm program between indicators, depending on the process details reflected and the emission factors attached to them. To analyze the accuracy of an indicator, its emission estimates are aggregated over all simulated farms and reduction steps, and then compared to the results for the reference indicator. Figure 2 shows how the resulting deviations depend on the milk yield of the initial herd. The vertical lines show the relatively limited minimum and maximum variation in GHG accounting accuracy for differentially-sized farms with identical milk yields. Economies of scale in investment decisions that change, for example, the type of manure storage or application technique obviously trigger some differences in realized abatement options between different sized farms; however, they have little impact on accuracy.

Figure 2: **GHG accounting accuracy of indicator schemes** (average % differences compared to the benchmark indicator. Variations due to herd size changes are shown by the vertical lines)



Source: own calculation and illustration.

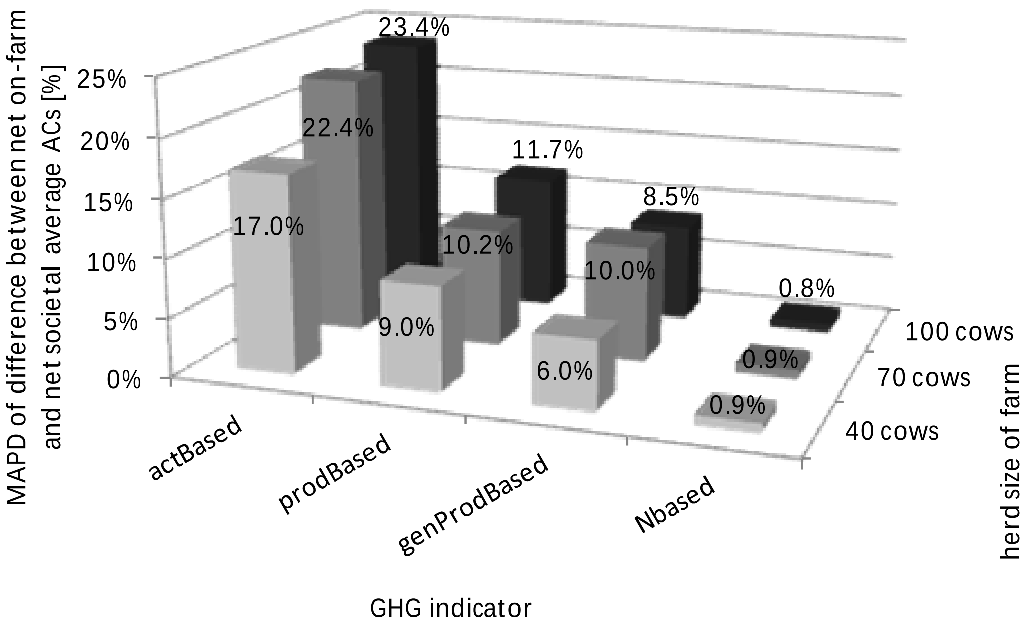
With the reference indicator assumed to be the best proxy for real emissions, the quite similar *NBased* indicator has the expected highest accuracy with a mean absolute percentage deviation (MAPD) of 1.1% over all simulation runs. The simple *actBased* indicator (MAPD 14.3%), with its fixed per animal factors, overestimates emissions for low milk yields up to around 6,500 kg per cow and year. The subsequent underestimation increases for output levels up to 10,000 kg as the default per cow emission parameter is calibrated on a cow with 6,000 kg milk output per year (IPCC 2006). The *prodBased* indicator (MAPD 8.5%) shows increasing overestimation with increasing milk output levels, whereas the *genProdBased* scheme (MAPD 6.4%) approaches the benchmark emissions from above with a higher milk yield, and even underestimates reference emissions by about 6% for a farm with cows that produce 10,000 kg per year.

Hence, the more sensitive an indicator reacts to changes in farm-level processes impacting GHG emissions, the more accurate are the resulting GHG inventories.

7.4.2 Induced abatement costs

When emission ceilings are implemented, the different indicators each trigger specific changes in farm plans and related profit losses, leading to markedly different ACs and MACs. Furthermore, the underestimation or overestimation of actual emissions determines differences between net on-farm and net societal abatement costs. Figure 3 illustrates the MAPD for differences in average ACs derived over all simulated farm runs and reduction steps (illustrated for three initial herd sizes).

Figure 3: **Mean absolute percentage difference between net on-farm and net societal average ACs** (net societal average ACs as base for calculation)



Source: own calculation and illustration.

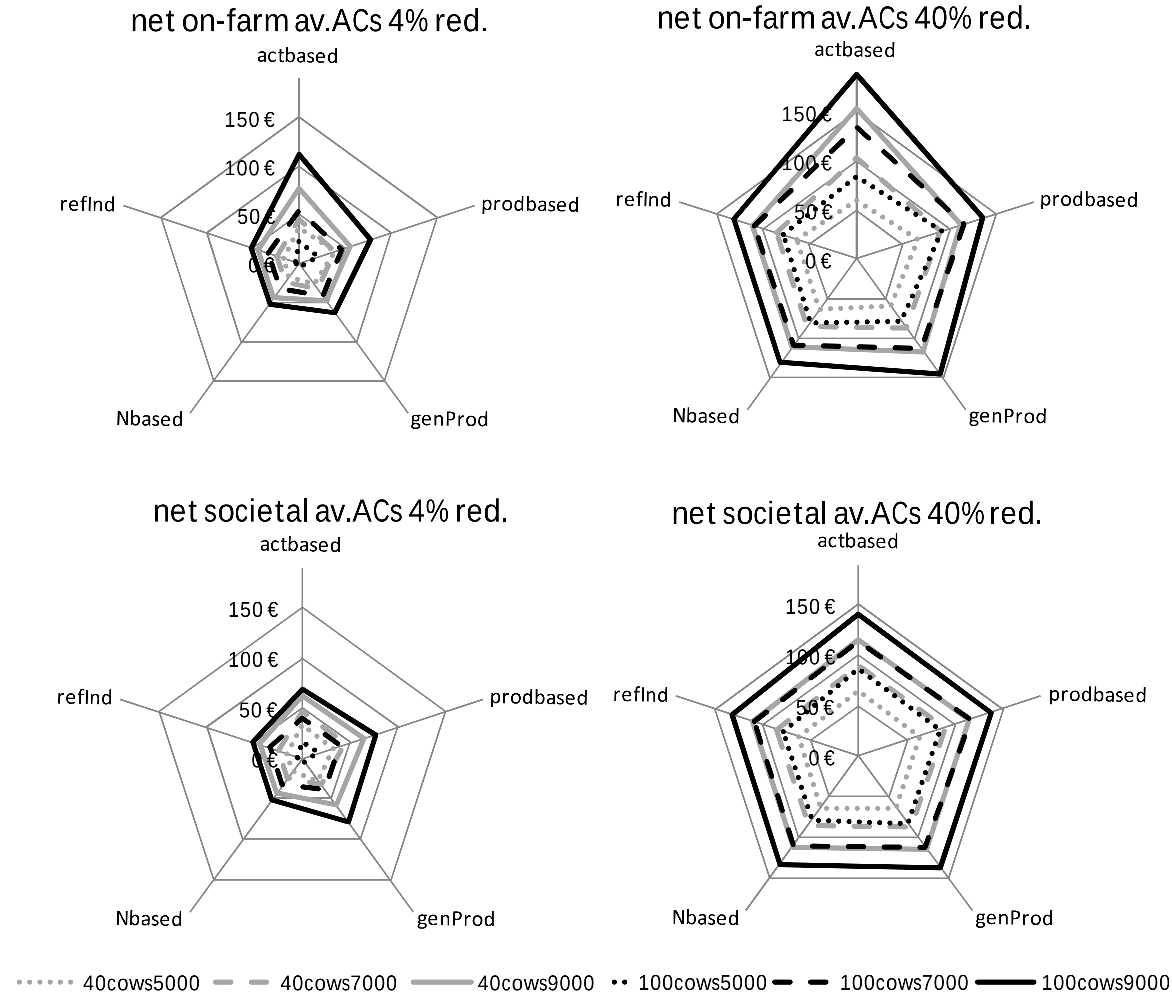
More detailed and thus more accurate indicators also show smaller differences between net on-farm and actual (net societal) ACs. Additionally, simpler indicators might not credit promising abatement options, and thus drive up ACs.

We illustrate selected findings for net on-farm and societal ACs in figure 4 by using simulation results of farms with initial herd sizes of 40 and 100 cows, and three milk yield levels (5,000, 7,000, and 9,000 kg milk per cow per year). For each indicator, the left-hand figures illustrate the average net on-farm and net societal ACs for a 4% reduction level of baseline GHGs; the right-hand figures illustrate a 40% reduction level. As expected, average ACs at 40% are higher as MACs increase in abatement quantities.

Average net on-farm ACs are highest under the simplest indicator (actBased): up to 112 €/t CO_2-equ. for a 4% level, and up to 188 €/t CO_2-equ. for a 40% reduction, which are far above current prices in the European ETS. However, the simpler indicators do not credit low-cost abatement options (see table 1), and thus provoke higher ACs. The most detailed indicator scheme (*refInd*) leads, as expected, to the lowest average ACs, but even at a low 4% abatement level, its ACs are already above current permit prices for the illustrated farms.

Independent from the chosen indicator, farm attributes show a sizeable impact on mitigation costs. For example, average net on-farm ACs vary between 64 and 131 €/t CO_2-equ., depending on farm attributes under the *NBased* indicators at a 40% GHG reduction.

Figure 4: **Net on-farm and net societal average ACs for a 4% and 40% reduction of baseline GHGs depending on indicator scheme [€ per ton CO_2-equ.]**



Source: own calculation and illustration.

The two spider charts in the lower part of figure 4 show that differences in net societal ACs between indicators are considerably smaller than for net on-farm ACs. Normalization by actual emitted GHGs reduces the ACs for simpler indicators, which shows that these indicators underestimate actual abatement on the example farms. At the 4% reduction requirement, high differences in average ACs between indicators can be observed even after normalization. These cost differences vanish almost completely at a 40% GHG reduction level, as abatement strategies no longer differ between indicators at higher abatement levels. The advantage in ACs of more detailed indicators thus vanishes with increasing abatement levels.

Figure 4 compares average ACs only for very low and quite high reduction requirements. We will now examine how the abatement level impacts average ACs (table 2 and figure 5), using two rather extreme examples from our simulations: a farm with 40 cows and 5,000 kg milk yield, and a farm with 100 cows and a milk yield potential of 9,000 kg in the initial herd.

As already seen from figure 4, at low reduction levels, MACs induced by less-detailed GHG indicators quite exceed those of the more detailed ones. With increasing GHG reduction, differences in MACs between indicators decrease, and hence the average ACs per kg CO_2-equ. also align (dividing the integral below the MAC curve by the GHG abatement amount). Table 2 depicts the absolute values of average net on-farm and net societal ACs for the two example farms depending on the chosen indicator scheme. Cases where farmers' costs per unit exceed the costs per actual unit abated appear in bold.

Table 2: **Average ACs of different specified farms and reduction levels** (bold numbers in the lower tables indicate a reduction of derived average ACs by normalization to net societal level)

40 cows with milk yield of 5,000 kg/cow and year							
net on-farm av.ACs [€/t CO_2-equ.]							
% red.	4%	10%	16%	22%	28%	34%	40%
actBased	37.3	45	49.5	52.5	58	62.4	60.3
prodBased	39.6	48.3	53.6	58.3	63.1	68.3	66.1
genProdBased	25.7	39	44.2	47.6	52.1	57.5	58.5
Nbased	16.5	35.2	43.7	48.8	53.4	59.7	64.2
refInd	15.6	34.7	43.5	48.2	52.3	58.5	63
net societal av.ACs [€/t CO2-equ.]							
% red.	4%	10%	16%	22%	28%	34%	40%
actBased	34	43.3	48.9	52.3	58.3	64	63.8
prodBased	32.2	42.5	48.1	55.9	58	63.9	63.4
genProdBased	28.2	41.5	47.2	50.9	56	61.7	63
Nbased	16.4	34.8	43.2	48.1	52.7	58.8	63.1
refInd	15.6	34.7	43.5	48.2	52.3	58.5	63

100 cows with milk yield of 9,000 kg/cow and year							
net on-farm av.ACs [€/t CO_2-equ.]							
% red.	4%	10%	16%	22%	28%	34%	40%
actBased	112.3	145.2	166.6	175	180.2	184.8	187.6
prodBased	77	106.1	119.6	126.7	130.3	133.4	135.7
genProdBased	62.2	99.5	119.4	130.7	136.2	140.8	144.4
Nbased	51.6	86.3	106.2	117.6	123.3	127.4	131
refInd	51.2	86.4	106.8	117.5	124.2	128.2	131.8
net societal av.ACs [€/t CO2-equ.]							
% red.	4%	10%	16%	22%	28%	34%	40%
actBased	68.8	138.6	130.4	137.4	136.4	141.6	139.7
prodBased	76.6	115.3	127.3	133.4	136.2	140.1	138
genProdBased	78.5	106.8	128.4	129.2	137.9	141.5	137.5
Nbased	51.8	86.7	106.9	118.3	124	128.1	131.8
refInd	51.2	86.4	106.8	117.5	124.2	128.2	131.8

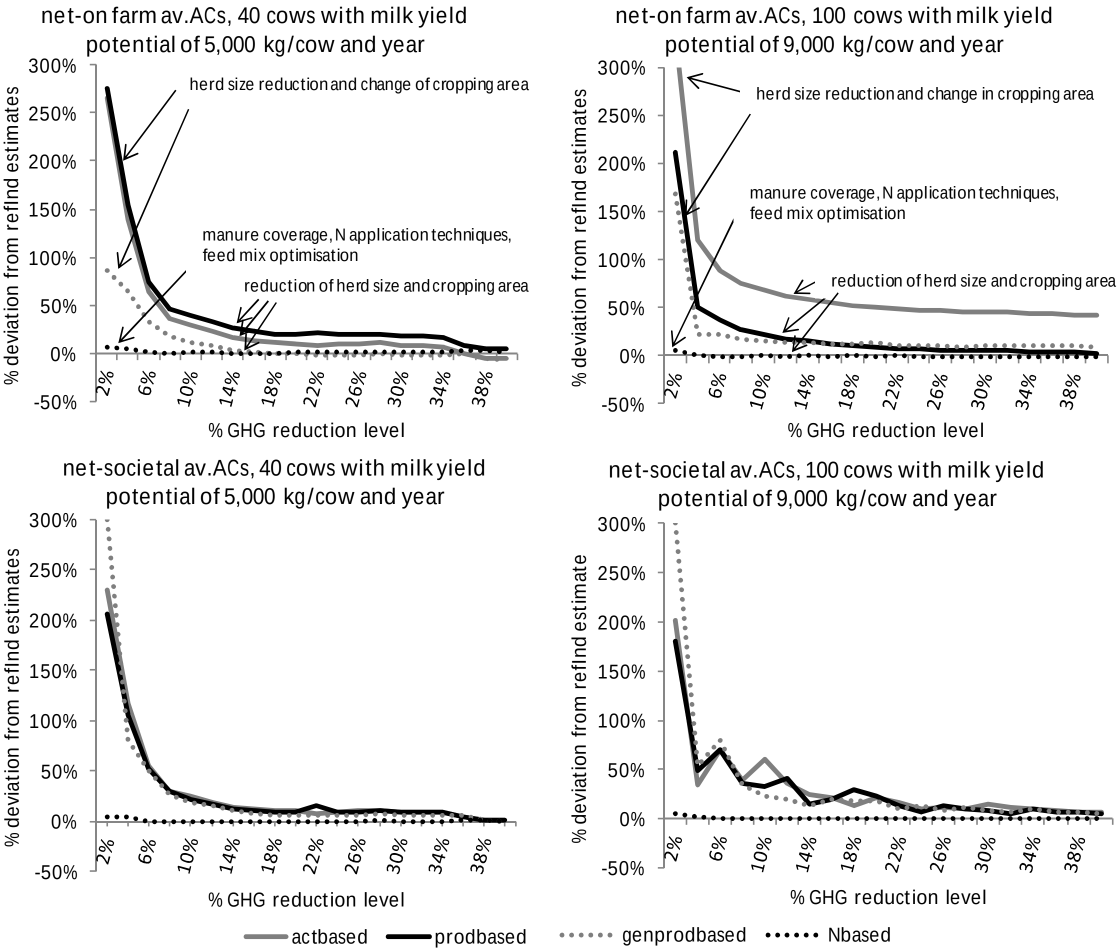
Source: own calculation and illustration.

The derived costs for both farms differ markedly in absolute level. Additionally, the bias between average on-farm and societal ACs can be quite high and increases as herd size and simplicity of the indicator increase, as also illustrated by the MAPD shown in figure 3.

Figure 5 illustrates this for the different indicators by showing the percentage differences of average ACs to the reference indicator, depending on the GHG ceiling.

The upper part of figure 5 illustrates the differences between average net on-farm ACs. As expected, especially for the first reduction steps, average net on-farm ACs induced by the *actBased* indicator by far exceed those under the reference indicator. For the larger, higher milk-yielding farm shown on the right-hand side, these differences remain quite high also for higher abatement efforts. However, average costs for a 20% emission reduction under the *actBased* indicator for the smaller, less-intensive farm type shown on the left-hand side are still 11% higher than for the reference indicator. This again highlights that simple indicators might provoke high on-farm GHG reduction costs, independent of the reduction level.

Figure 5: **Net on-farm and net societal average ACs for a reduction of baseline GHGs depending on indicator scheme, relative to refInd estimates**



Source: own illustration.

The two lower graphics compare the net societal average ACs. At low percentage reduction amounts, all simple GHG accounting schemes induce very high and similar cost differences compared to the reference indicator (table 2). For the illustrated 100-cow farm, the average net societal ACs induced by the *actBased* indicator for a 2% reduction (57.7 €/t) are more than three times higher than for the reference indicator (19.1 €/t). For higher reduction levels, the cost disadvantage of simple indicators drops: average net societal ACs align with increasing abatement as similar abatement strategies are chosen.

The cost differences between indicators reflect the abatement strategies credited by the indicators and chosen by the farmer. Although indicators account for various mitigation options, many mitigation options have a maximum level of abatement potential. If the abatement potential of a cost-saving option is fully utilized, the next expensive option is applied. This leads to increasing MACs and, if all indicators credit the more expensive options such that farms do not exit completely, potentially to mitigation strategies for higher abatement levels with little difference between GHG indicators. At a 2% reduction level, under the detailed indicator (*NBased*), both displayed farms react by adjusting the animals' diet, manure removal frequency, manure coverage techniques, as well as by optimizing application time and quantity of organic and synthetic fertilizer (figure 5). However, the contribution of these measures differs between the two farms, and the possible contribution of the measures to overall GHG abatement may be limited. Under the simplest *actBased* indicator, only the herd size and the cropping pattern can be adjusted, which is far more costly. Once the mitigation potential of low-cost measures credited by more detailed indicators is exhausted, further abatement requires reductions in herd sizes and thus leads to similar reduction strategies across indicators at higher abatement level. Consequently, the advantage of more detailed indicators in offering more cost-saving mitigation options decreases with increasing abatement levels.

In general, for higher abatement levels, detailed indicators still show lower average net on-farm ACs. However, once ACs are normalized to net societal ACs, more detailed indicators lose a larger part of that advantage.

7.4.3 Feasibility of indicators

For relatively simple indicators, the necessary data on activity levels or production quantities for cropping[49] or animal husbandry[50] might be easy to collect and control, and are often already part of many farmers' legal reporting obligations. Information on specific feed ingredients in rations, the digestibility of feed supplements, and declarations of manure storage times, to provide some examples of the data necessary for complex

[49] Agricultural land registers.

[50] Since September 1999, all cattle in Germany must be reported to the HIT data pool following the § 24f Livestock Movement Order. The data pool is part of the "traceability- and information system for animals," which was implemented in all EU Member States following the EG-decree (EG) no. 820/97.

indicators, are currently not available and will be costly to monitor and difficult to control, which potentially lowers data reliability (OENEMA et al., 2004: p.175). Both farmers and policy-makers therefore request indicators that draw on reliable and available farm-level data that clearly favors simple indicators (e.g., *actBased*).

7.5 Discussion

Though our findings regarding the abatement costs per kg CO_2-equivalent cannot be directly compared to other studies for a number of reasons, we will highlight only the most important ones. Firstly, studies in the field publish MACs for different GHG abatement ceilings that might not match the range we analyzed. Secondly, we conducted simultaneous experiments over a wider range of relevant farm attributes without aiming for a consistent weighting that reflected the underlying farm population, whereas others studies might use farm types derived from the European Farm Accountancy network, where aggregation weights to the farm population are available (DECARA et al., 2005). A systematic analysis reflecting the actual distribution of farm attributes is a portion of planned future work. Thirdly, we abated GHGs over a longer decision horizon while allowing for certain investments, a feature not often found in other studies. And finally, the indicators used in this paper were often not easy to compare to other indicators. Nonetheless, the MACs values we simulated at higher abatement levels are in a similar range as findings, for example, from DECARA et al. (2005), DECARA and JAYET (2006), DURANDEAU et al. (2010), or PÉREZ (2005), but typically lower for detailed indicators and moderate abatement levels, which is not astonishing given our more detailed analysis.

We went a step beyond existing studies by comparing variously detailed GHG calculation schemes to examine the effect of the indicator's construction on the occurring GHG abatement efforts, costs occurring on the farm level, and possible biases and trade-offs that have to be accounted for in environmental policy discussions. In our context, where emissions cannot be measured directly, the findings underscore that typically quantitative analyses covering variously detailed indicators are necessary.

7.6 Summary and policy conclusion

This paper analyzed trade-offs between measurement accuracy, induced abatement cost levels, and the feasibility of five differently detailed GHG emission indicators for dairy farms based on simulations with a highly detailed bio-economic single-farm optimization model. Results show distinct differences concerning feasibility, measurement accuracy, and induced abatement cost between the indicators.

We found that only highly detailed GHG indicators credit the most promising (that is low-cost) abatement options and tally emissions accurately at different abatement levels and for farms differing in key attributes such as milk yields.[51] Simple indicators lead to strong overestimations and underestimations of actual emission reductions for certain farms, especially those with milk yields differing considerably from the values the simple indicators are calibrated on. When a policy-maker is confronted with very heterogeneous dairy farm structures, like in the EU or even in Germany itself (IT.NRW, 2012), highly detailed indicator schemes would be preferable to avoid inaccuracies in GHG inventory estimates, as well as an unfair treatment of different farm types; this point was also raised by CROSSON et al. (2011: p.41) and OENEMA et al. (2004: p.178). A lack of accuracy also leads to a difference between on-farm abatement costs (related to GHG emissions calculated by the indicator used) and societal abatement costs (related to the actual GHG emissions abated). Simpler indicators do not signal the actual (societal) costs per emission unit well, whereby in some cases it overestimates the societal costs and thus leads to the under-provision of abatement activities. On the other hand, detailed indicators provide farms with more accurate estimates of emissions and thus signal the societal costs of each abatement option more accurately way. The resulting difference between on-farm and societal costs lowers the precision of any indicator-based policy instrument and thus adds a further argument for more detailed indicators (see figure 3). Moreover, in all analyzed cases, detailed indicators show a cost advantage based on their ACs.

These advantages with respect to accuracy and ACs contrast with the obvious disadvantage of detailed indicators regarding data requirements. The necessary data on farm processes are currently not collected, would create quite some reporting burden for farmers, and once reported might be both challenging and expensive to maintain. Further on, the cost advantage of detailed indicators tends to level out at higher GHG abatements. That stems from the fact that low-cost abatement options only credited by detailed indicators have only a limited abatement potential. Still, differences in average net on-farm ACs do not vanish completely.

[51] See the mean absolute percentage deviations (MAPD) of GHG estimates from the benchmark scenario, i.e., figure 2.

From a societal point of view, a slightly different picture emerges once on-farm ACs are corrected by the GHG accounting bias of the indicators. More detailed indicators still show a pronounced cost advantage with regard to the resulting average net societal ACs at low abatement levels. However, that advantage vanishes once the potentials of low-cost measures are fully utilized, which happens already at relatively low GHG abatement levels. Accordingly, the advantage of detailed indicators diminishes with increasing mitigation. Once all indicators trigger comparably more expensive abatement strategies for each single farm modeled, average net societal ACs as calculated by the different indicators converge (see figure 5).

Thus, from a societal perspective, when higher emission abatement levels are targeted, less detailed and easier-to-monitor and easier-to-control indicators might be advantageous. Our findings suggests that the accuracy of quite simple indicators, which for example only relate to herd size and crop areas, can be improved if some relatively easy-to-monitor or easy-to-control farm attributes such as the milk yield or manure storage and distribution technology are taken into account.

Our results therefore indicate that the performance of an indicator with regard to accuracy, abatement costs and feasibility depends on the targeted range of emission reductions.

Our results have several implications for the implementation of dairy farms in EU GHG abatement efforts. Even with our quite detailed model approach and the most flexible and detailed indicator scheme, we find ACs that are considerably above current tradable permit prices in the EU ETS, which are at about 7 €/ton CO_2-equ. When becoming part of that ETS, dairy farms would thus rather buy permits instead of actually mitigating larger shares of their baseline GHGs. At these permit prices, our estimates hint at meager reductions between 2-4 % on the farm level (table 2), even for the most detailed indicator. But to achieve these reductions under market-based policy instruments, highly detailed process data would be necessary, leading to high monitoring costs. Similar to ANVEC (2011: p.111) we conclude that it would not be efficient to include dairy farms—at least under current conditions—into the ETS once the related transaction costs are considered. MARBEK RESOURCE CONSULTANTS (2004, cited in BETZ, 2006: p.8) estimate the administrative costs for agriculture to be in the range of 2.54-21.88 €/ton CO_2-equ., compared to 0.04-0.13 €/ton CO_2-equ. for other sectors, due to the small size of the individual mitigation projects at the farm level. These administrative costs alone could hence easily exceed current permit prices. Including dairy farms in the EU ETS would only become efficient in the case of a substantial increase in permit prices, or alternatively, the development of new, low-cost abatement options or high rates of technical progress.

To date, relatively few abatement options show ACs below permit prices, thus making them economically efficient. These options could more easily be implemented based on statutory requirements with low administrative costs. Specific GHG-reducing features of installations are promising, for example for manure storage, where monitoring costs might be quite low because existing laws typically already require the inspection of any newly-erected constructions. However, the resulting abatements are quite low, probably ranging from 2-4%.

Our analysis thus clearly suggests that a market-based policy to abate GHGs from dairy farms is currently not an efficient option. The analysis also highlights that indicator accuracy is not only a technical detail, but one that introduces differences between private and societal ACs and thus can harm the economic efficiency of market-based instruments, while at the same time provoking the unfair treatment of agents. More generally, the analysis underlines that policies targeting externalities, which cannot be measured directly and thus require indicators, demand a detailed analysis of potential indicators with regard to accuracy, abatement cost levels, and feasibility at different target levels. From our conceptual analysis and quantitative findings we conclude that the choice of indicator, policy instrument and the targeted abatement level are closely interlinked and are not separable aspects of environmental policy design.

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Appendix

Appendix 1: Economic explanation of the modeling approach (based on DAIRYDYN from LENGERS and BRITZ 2012)

Objective function:

(1) $Max!\,\pi = l_t * \left(1 - \frac{d}{100}\right)^t / t$

(1.1) s. t. $\sum_{k=1}^{n} a_{e,k} x_k \leq b_e$; $k = 1 \dots n; x \in \mathbb{Z} \cup \mathbb{R}$

(1.2) and $X_k \geq 0$

π = average of the net present value of accumulated liquidity

l_t = accumulated liquidity of the firm in last year t

d = private discount factor (e.g. 2% -> d = 2)

t = time index for years of planning horizon

$a_{e,k}$ = requirement of one unit of activity k on the resource of type e

x_k = amount of activity k

b_e = maximum of resource e available

Emission constraint for stricter emission ceiling:

(1.4) $\sum_k ef_{j,k} x_k \leq (1 - \alpha_i)\varepsilon_{0,j}$

i = step of simulation (reduction step compared to baseline)

α_i = amount of total percentage reduction of emission in step i compared to baseline

$\varepsilon_{0,j}$ = baseline emission of the farm without emission restriction

$ef_{j,k}$ = total amount of GHG emissions related to one unit of activity k under indicator j

j = index for specific indicator

Derivation of ACs:

(2) $AC_{i,j} = \pi_{0,j} - \pi_{i,j}$

$AC_{i,j}$ = total abatement costs in simulation step i, using indicator j

$\pi_{i,j}$ = value of objective function in simulation step i, using indicator j

$\pi_{0,j}$ = value of objective function in the baseline, using indicator j

Derivation of average ACs:

(2.1) $av.ACs_{ij} = \frac{\pi_{0,j} - \pi_{i,j}}{\varepsilon_{0,j} - \varepsilon_{i,j}}$

$av.ACs_{i,j}$ = average abatement costs for the total reduction of step i, using the indicator j

$\varepsilon_{i,j}$ = emission amount in simulation step i, calculated with indicator j

Normalization to actual abated emissions:

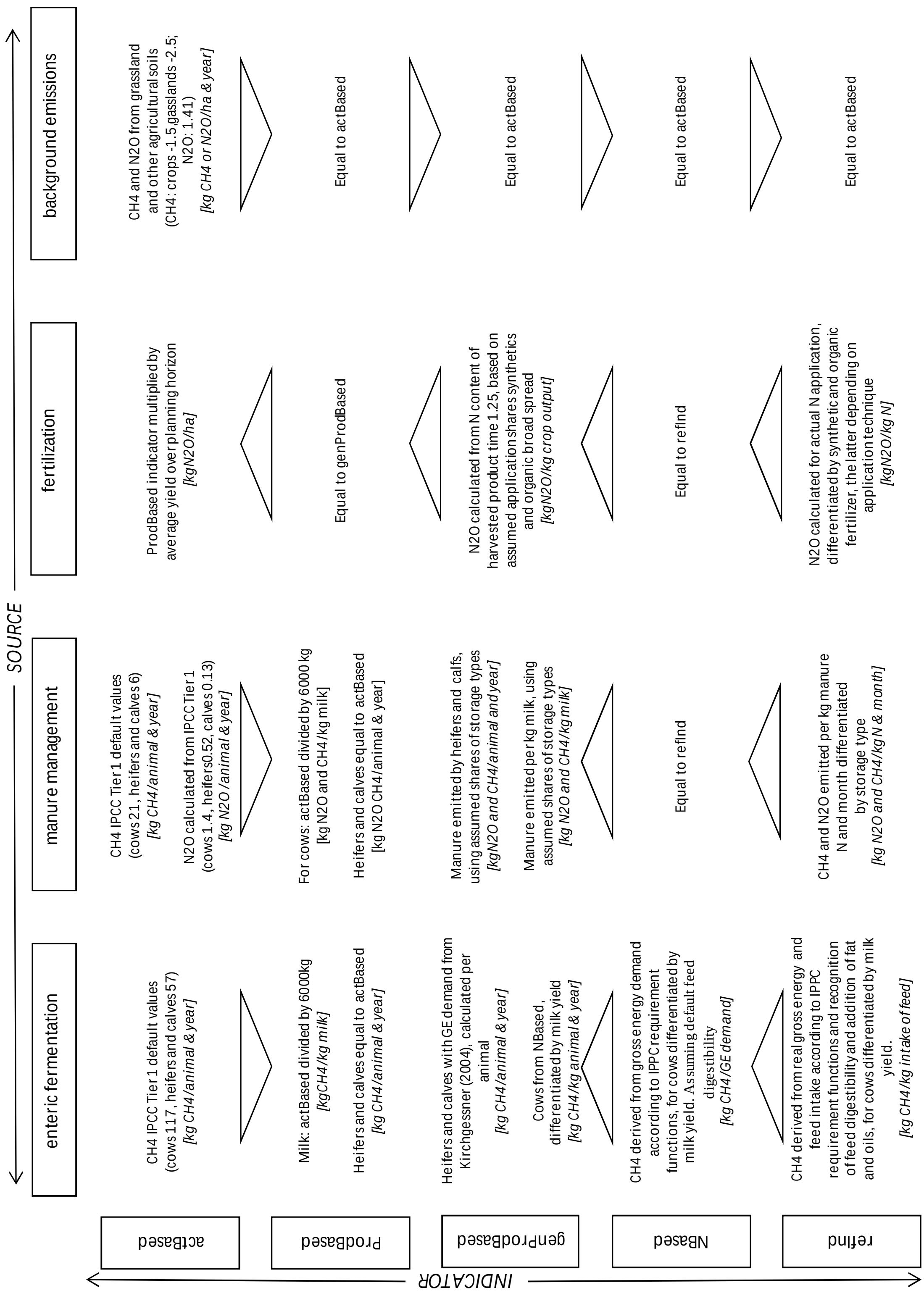
(2.2) $av.ACs_{i,j}^{norm} = \frac{\pi_{0,j} - \pi_{i,j}}{\varepsilon_{0,r} - \varepsilon_{i,r}}$

$av.ACs_{i,j}^{norm}$ = average abatement costs for the total reduction of step i, using the indicator j

$\varepsilon_{i,r}$ = emission amount in simulation step i, calculated with reference indicator r

r = index for reference indicator

Appendix 2: Indicator schemes



Source: own illustration.

Chapter 8: **What drives marginal abatement costs of greenhouse gases on dairy farms? A meta-modeling approach**[52]

Abstract

This paper discusses the quantitative relations between the marginal abatement costs (MACs) of greenhouse gas (GHG) emissions on dairy farms and factors such as herd size, milk yield and available farm labor, on the one hand, and prices, GHG indicators and GHG reduction levels, on the other. A two-stage Heckman procedure estimates these relations from a systematically designed set of simulations with a highly detailed mixed integer bio-economic farm level model. The resulting meta-model is then used to analyze how the above mentioned factors impact abatement costs. We find that simpler GHG indicators lead to significantly higher MACs, and that MACs strongly increase beyond a 2 to 5% emission reduction, depending on farm attributes and the chosen indicator. Economics of scale let MACs decrease rapidly in farm size, but the effect levels off beyond a herd size of 40 cows. As was to be expected, the main factors driving gross margins per dairy cow also significantly influence mitigation costs. The results thus suggest high variability of MACs on real life farms. In contrast to the time consuming simulations with the complex mixed integer bio-economic programming model, the meta-models allow the distribution of MACs in a farm population to be efficiently derived and thus could be used to upscale them to regional or sector level.

Keywords: *emission indicators, meta-modeling, Latin-Hypercube sampling, marginal abatement costs, dairy farms.*

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[52] This is the pre-peer reviewed version of the following article: LENGERS, B., BRITZ, W. and K. HOLM-MÜLLER: What drives marginal abatement costs of greenhouse gases on dairy farms? A meta-modeling approach, *Journal of Agricultural Economics.* A revised version of this chapter is accepted for publication and the final version is available at onlinelibrary.wiley.com. The research was funded by a grant from the German Science Foundation (DFG) with the reference number HO 3780/2-1. A former version of this study was presented at the 2013 AURÖ-workshop in Frankfurt (Oder), Germany.

8.1 Introduction

The mitigation of greenhouse gases from agricultural production processes is broadly discussed from a political and a scientific viewpoint, both with regard to abatement costs (e.g. ANVEC, 2011; DECARA and VERMONT, 2011) and whether and how to incorporate agriculture into GHG reduction efforts (e.g. RAMILAN et al., 2011). Whereas the parties to the Kyoto protocol have to report agricultural GHG emissions as part of their GHG inventories, to our knowledge only New Zealand so far plans to incorporate agricultural GHG emissions. Specifically, the New Zealand Emission Trading Scheme (ETS) indirectly allocates GHGs of agricultural production (but up to now only for reporting purposes) to meat and milk processors, fertilizer companies and live animal exporters who are already ETS participants (MPI, 2013). Any decision on policy instruments and potential reduction levels requires knowledge about possible abatement options, abatement potentials and, in particular, related mitigation costs.

Various studies are available regarding possible abatement strategies and potential GHG savings from agriculture (e.g. FLACHOWSKY and BRADE, 2007; NIGGLI et al., 2009; OLESEN et al., 2004, 2006; SMITH et al., 2008), and related abatement and marginal abatement costs (MACs). However, these studies often concern specific regions or single farm types (DECARA and JAYET, 2000; DURANDEAU et al., 2010; GOLUB et al., 2009; LENGERS and BRITZ, 2012; MORAN et al., 2009; PÉREZ, 2006; RAMILAN et al., 2011) and show a high variability in the estimated MACs. In a recent review, VERMONT and DECARA (2010) concluded that the observed variability in MAC estimates is rooted to a large extent in methodological differences in the studies.[53] More generally, BARKER et al. (2002) and KUIK et al. (2009) showed that MACs derived by simulation models depend on their structural characteristics and further assumptions such as the emission baseline used and the relevant time interval for emission quantification. In addition, LENGERS et al. (2013a) stress the importance of the chosen abatement level, and together with LENGERS and BRITZ (2012) highlight the importance of the GHG accounting scheme for the estimated MACs, a point often neglected in other studies. They argue that the chosen abatement strategies and consequently MACs strongly depend on the GHG accounting scheme as farmers only adopt options which are credited. Besides that, an economic perspective suggests that

[53] Differences in the used model approaches (engineering model, supply side approach, equilibrium model).

MACs should clearly depend on prices of input and output and further characteristics of the farms investigated, points often not closely analyzed in existing studies.

To summarize, available studies offer a wider range of MAC estimates which are hard to generalize as they depend on factors not systematically controlled for, such as exogenous assumptions on prices, the GHG indicator used, farm characteristics or the model structure. Published meta-analyses on MACs by VERMONT and DECARA (2010) as well as KUIK et al. (2009) only investigate differences in methodological approaches and assumptions. To our knowledge, no study exists which systematically analyses drivers of differences in MACs and thus farm income changes provoked by GHG related policy instruments, which is certainly a hotly discussed aspect in policy debate concerning inclusion of agriculture in emission reduction efforts. The contribution of the present paper is twofold in this respect: proposing a methodology for meta-analysis and applying it to German dairy farm conditions to get insight into the above mentioned relations.

Given the quite heterogeneous dairy farm structures in Germany, the evidence, e.g. from LENGERS and BRITZ (2012), with regard to differences in MACs and the necessity of GHG reduction policies to use emission indicators as technical control parameters (SCHEELE et al., 1993: p.298), two questions arise: (1) which are the most important farm characteristics impacting the MACs and what is their quantitative effect? (2) What is the relation between the applied GHG indicator and other drivers such as prices and the MACs?

Answering these questions requires a set of single farm observations of MACs with sufficient variation in key factors. But time and cost considerations exclude real world experiments to derive MACs for a larger group of farms. Hence, different computer based simulation models are used instead to estimate GHG abatement costs in agricultural production systems (e.g. CAPRI by PÉREZ (2006) ModelFarm by WEISKE and MICHEL (2007), AROPAj by DECARA and JAYET (2000) and DECARA et al. (2005), DAIRYDYN by LENGERS and BRITZ (2012)).

In order to reflect differences related to farm attributes and GHG indicator choice, single farm approaches are best suited as they depict the complex bio-physical and bio-economic processes in agricultural production in sufficient detail.

However, findings for a specific single farm can hardly be generalized to more general farm types or regions (STOKER, 1993), the level of interest for policy decisions. In order to cover the distribution of relevant factors in the farm population, a large set of computer experiments is needed. As each experiment requires a complete simulation for a single farm runtime considerations become important if they restrict the amount of possible experiments. (BOUZAHER et al., 1993: p.3; CARRIQUIRY et al., 1998) For example, simulating MACs for just 70 dairy farms over a 15-year planning horizon with the DAIRYDYN model used in LENGERS et al. (2013a) took more than three days with an 8-core processor.

A more efficient way to conduct single farm simulations is hence necessary. Meta-modeling seems inviting as it can replace simulations using a complex computer model with a far simpler one, helping to overcome computational restrictions. A meta-model approximates the output (response) of the more complex model using standard statistical techniques on model results from representative variations of determinants in the underlying complex model. It identifies the most important factors for model results and leads to a simpler functional form with fewer input variables (factors) (KLEIJNEN, 2009: p.707). (CARRIQUIRY et al., 1998: p.507) A meta-model quantifies major input-output (I/O) relations embedded in the structure of the more complex model (KLEIJNEN, 2008) and can thus improve our understanding of real-life systems (BOUZAHER et al., 1993: p.3).

Consequently, the first step in the development of a meta-model is the generation of a set of model results with representative variations of determinants. Here, efficient sampling algorithms are helpful which allow for relatively compact sets of experiments which are still representative of the factors' distributions in the population.

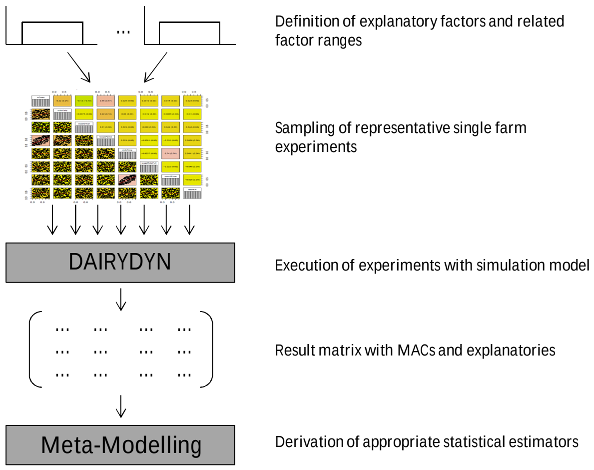
Accordingly, in this paper we use an appropriate meta-modeling procedure and apply it to a real-life example. The meta-model can summarize the behavior of the underlying, more complex simulation model for a sample of simulated farms, in our case focusing on the dairy sector. The meta-model allows us to analyze the relations between MACs and the predefined characteristics of dairy farms, prices and process variables. This will improve our understanding of how MACs depend on these factors and the applied GHG indicator scheme in a larger sample of farms, complementing existing literature which typically provides results either for selected single farms only, or at a rather high aggregation level of larger administrative regions, but cannot give farm specific information on what drives MACs.

The organization of the paper is as follows. First, we briefly outline DAIRYDYN, a highly detailed bio-economic model for specialized dairy farms, which we use to create a representative sample of results to derive statistical meta-models. Next, we present the different GHG indicator schemes for which MACs will be investigated. The following sections discuss the set up of our experiments which cover relevant dairy production systems in North Rhine Westphalia, Germany, based on an efficient space filling sampling procedure under recognition of factor correlations. Subsequently, we discuss the estimation of the meta-models based on a Heckman two-stage selection procedure. The statistical estimator delivers both test statistics and error terms for the given sample and shows its validity regarding the sampling background. Simulations with the meta-models are used to present the main findings before we summarize and conclude.

8.2 Material and methods

Long lasting and expensive real-life experiments would be necessary to determine MACs on actual dairy farms, which is impossible for a large sample. Thus, we employ instead "what-if simulations" following DOE-principles (design of experiments) (KLEIJNEN, 1999; KLEIJNEN, 2005; KLEIJNEN et al., 2005) using the single farm model DAIRYDYN to generate outputs of a representative sample.[54] In order to account for the interactions of different factors[55] such as farm attributes, prices and indicator choice on the MACs, we construct a larger set of experiments where factor levels of implemented variables are systematically and simultaneously[56] varied, also respecting possible correlations. Simulation results are then used to derive appropriate statistical meta-models. The systematic of the overall approach is visualized in the following figure.

Figure 1: **Proceeding of our analysis**



Source: own illustration.

[54] Random permutation of factor levels so that design matrix still ensures orthogonality (KLEIJNEN et al., 2005: p.278).

[55] In the realm of DOE simulation an input variable or parameter is understood as a factor. Factors can be of qualitative, quantitative as well as discrete or continuous character. (KLEIJNEN et al., 2005: p.264)

[56] Only changing one factor at a time is not the accurate scientific way to analyse effects of this factor because single factor effects may be different through interaction when other factors change simultaneously (KLEIJNEN, 1999).

8.2.1 The simulation model DAIRYDYN

DAIRYDYN (LENGERS and BRITZ, 2012) is a highly detailed mixed integer linear programming model for the simulation of economically optimal farm-level plans on specialized dairy farms, realized in GAMS (General Algebraic Modelling System). It maximizes the expected net-present value over several years and different states of natures under GHG emission ceilings related to specific GHG indicators. It takes into account investment and labor use decisions, respecting their non-continuous character, and considers sunk costs of past and future investment decisions along with the evolving path dependencies. DAIRYDYN is based on a detailed production based approach, simulating farm management decision and related material flows for animal husbandry, cultivation of land, feed production and feeding as well as manure management, partly on a monthly basis. The model is designed to analyze impacts of GHG emission ceilings compared with a so-called baseline run which comprises an optimized farm plan without any emission ceiling.

The decision variables are linked to a GHG accounting module which incorporates five different IPCC (2006) (Intergovernmental Panel on Climate Change) based GHG calculation schemes to quantify the farm-level GHG inventory (LENGERS, 2012). These indicators differ in the level of detail of required process information. In our analysis, we only use the two extreme indicators, i.e. the simplest (actBased) based on default per animal or per ha emission parameters (comparable to IPCC Tier 1 approach) and the most detailed (refInd), which implements highly detailed process information in its GHG calculation as, for example, milk yield level, composition of the ration, type and duration of manure storage, synthetic and organic fertilizer practice and manure coverage techniques, at a yearly and where reasonable monthly resolution (derived from the IPCC Tier 3 approach) (see LENGERS and BRITZ (2012) and LENGERS et al. (2013a) for more detail or LENGERS (2012) for a full description). We leave the remaining three intermediate indicators out as earlier results showed that abatement strategies and related costs for these intermediate indicators do not differ significantly from one of the two extremes (LENGERS and BRITZ, 2012; LENGERS et al., 2013a). Whereas the actBased indicator is only sensitive to changes in activity levels (change of animal numbers, cropping and grassland acreage decisions), the refInd indicator also accounts for changes in the intensity of management, feed composition, fertilizer practice and manure storage types and time (LENGERS, 2012). This leads to different abatement strategies and adherent costs under the indictors as shown by LENGERS and BRITZ (2012).

The model generates MACs by relating marginal profit losses to changes in the GHG emission ceiling (for a more detailed description see LENGERS and BRITZ, 2012: p.131). These MACs are termed net on-farm because they exclude transaction costs relating to administrative and control efforts.

So far, DAIRYDYN has been used in the studies of LENGERS and BRITZ (2012) and LENGERS et al. (2013a) for the estimation of abatement strategies and related costs under different indicators and in a paper of LENGERS et al. (2013b, see chapter 6 of this dissertation) to compare modeled GHG estimates with real-life long term measurements. The first two studies report MACs in the range of estimates from other comparable studies, whereas the third shows a rather good fit for the estimated GHGs to real-life examination results of an experimental dairy farm installation in North Rhine Westphalia, Germany.

8.2.2 Sampling procedure

We aim to generate a sample of computer experiments which is representative of the dairy farm population in the region of North Rhine Westphalia, Germany. Key farm attributes such as milk and crop yields, herd sizes, stocking rates, labor productivity, production costs and milk prices are chosen to reflect their observed ranges in the farm population. KLEIJNEN (2005: p.290) names the ranges of possible factor and value combinations in such computer experiments "the domain of admissible scenarios". Even if we define for each factor only a limited number of possible levels (e.g. 30, 60, 90, 120, 180 cows), it would still be impossible to simulate all potential permutations of factor level combinations with DAIRYDYN. Therefore, a limited but representative set of factor level combinations is selected based on DOE (KLEIJNEN, 1999). Specifically, we apply a so-called Latin-hypercube sampling (LHS) method, which is more efficient than simple random sampling (GIUNTA et al., 2003: p.7; IMAN and CONOVER, 1980; IMAN et al., 1981: pp.176-177; IMAN, 2008; MCKAY et al., 1979). LHS defines a number of experiments which simultaneously change levels of various factors, while being representative for the full range of possible factor level permutations. Based on so-called space filling designs, LHS smoothly samples over the *k*-dimensional input space of the computer model for a defined size of the sample (IMAN et al., 1981: p.176; OWEN, 1992: pp.443-445). It does not necessitate a decision beforehand for which factors a more fine-grained resolution of levels is appropriate (IMAN, 2008). However, standard LHS assumes zero correlation between the factors which may lead to invalid statistics compiled from the output if factors are correlated in reality (IMAN and CONOVER, 1982: p.331). We thus employ a LHS procedure according to IMAN and CONOVER (1982) which considers factor correlations.

8.2.3 Explanatory variables

Keeping in mind that we want to investigate key attributes impacting MACs, two types of factors are potential candidates for our experiments: (1) economic drivers and farm attributes for which population statistics are available (BETTONVIL and KLEIJNEN, 1996; SÄRNDAL et al., 1992 cited in CARRIQUIRY et al., 1998: p.507) and/or (2) factors that are not conditioned by farm attributes, but which possibly describe a potential GHG reduction policy in place and allow conclusions about impacts that a design of GHG control policies may have on the resulting mitigation costs. We have chosen the following factors (the number in brackets indicates the factor class):

- **Starting herd size (1)**: This factor gives a good indication of the size of the farm. In DAIRYDYN it steers, for example, the initial endowment of stables, machinery and land. The farm size, via returns-to-scale, impacts production costs and should therefore impact the MACs.
- **Milk yield (1)**: The average milk yield per cow in the herd indicates the intensity of the production system: higher milk yield increases GHG emission per cow, but decreases emission per kg of milk produced. Besides its impact on production costs, the milk yield may therefore significantly impact abatement strategies and related costs under different indicators. (LENGERS and BRITZ, 2012)
- **Milk Price (1)**: The milk price predominantly impacts the revenues of the overall farm. Higher milk prices drive up the gross margin of a single cow and thus the cost of herd size reductions or a complete farm exit. Furthermore, it determines the optimal intensity of milk production where the marginal costs per unit of output are equal to the price.
- **Concentrate Price (1)**: The most important feed ingredient to control the energy level of the ration and hence the intensity level of the cows are concentrates. They are an important cost factor for mitigation options based on fodder optimization and more generally for the profitability of the farm.
- **Wage Rate (1)**: The wage rate is included in our simulations to analyze the impact of the opportunity costs of labor which impact farm size reductions

or a possible exit decision of a farmer if marginal returns to on-farm labor drop below the wage rate in response to an environmental restriction. More generally again, it impacts the overall profitability of the operation.

- **Working hours per cow and year (1)**: As the amount of labor available for one cow crucially determines the labor productivity of the farm, it is included for similar reasons as the wage rate.
- **Age of stables (1)**: The older the stables are, the earlier new investments in stables must be made to maintain the farm. New investments will also allow for an expansion strategy by increasing stable sizes. The stable age also clearly impacts the share of sunk cost over the simulation horizon if new investments are made in response to a GHG policy.
- **Time horizon (2)**: As we require a certain reduction of GHG only in average over the full simulation period, and not in each single year, a longer simulation period allows larger shifts of emission between years and increases the flexibility of the adjustment further as investment based mitigation measures face a longer depreciation time. This factor is an important aspect describing the reaction scope offered to the GHG regulated farms.
- **GHG restriction (2)**: The required GHG reduction clearly should impact MACs.
- **Indicator (2)**: LENGERS and BRITZ (2012) as well as LENGERS et al. (2013a) show that the indicator choice has a significant impact on the MACs as different indicators account for different sets of abatement strategies.

Admissible ranges for the factors (table 1) are taken from regional as well as country specific statistical data sources (BMELV, 1991-2011; BMELV, 2012; FDZ, 2013; IT.NRW, 2012; KTBL, 2010; LKV-NRW, 2012; LWK-NRW, 2008-2012) to ensure that the designed experiments fit to the actual population of dairy farms in North Rhine Westphalia, Germany.

Table 1: **Factor ranges of variables changed for DOE**

variable/ attribute	Unit	Min	Max	Data source
nCows^	Head	20	250	IT.NRW, 2012; LKV-NRW, 2012
Labor productivity°	Working hours cow^{-1} a^{-1}	30	72	LWK-NRW, 2008-2012; FDZ, 2013
Milk yield°	kg ECM cow^{-1} a^{-1}	5,000	11,000	LWK-NRW, 2008-2012; LKV-NRW, 2012
Construction year of stables	Year	1995	2005	
simulation horizon	Years	10	20	
Milk price	€-cent kg ECM^{-1}	25.97	36.44	BMELV, 1991-2012
Price concentrate	€ ton^{-1}	160	230	adapted to KTBL, 2010; LFL, 2012; LWK-NRW, 2008-2012
Wage rate	€ $hour^{-1}$	6	15	LFL, 2012; LWK-NRW, 2012

Prices are declared as yearly average prices. Values with ° are derived from 96% of the original data sets due to exclusion of extreme values. ^ denotes that farms with a herd size below 20 are excluded because it is assumed that they represent predominantly tethering houses; the simulation model only covers free stalls. Nevertheless, the remaining population still represents above 97% of the whole 2012 cow population in North Rhine Westphalia.

Source: own illustration following named references.

For a robust sampling, correlations[57] between variables are derived from different data sets like BMELV (several years), FDZ (2013), KTBL (2010: p.541), LWK-NRW (2008-2012) and the LKV-NRW (2012). We quantified the following correlations: between milk yield and herd size at 0.24*** and between milk yield and labor productivity at 0.18*, between milk price and concentrate prices of 0.76***, and between labor intensity per cow and herd size of -0.65***. Furthermore, though there is no statistical data on this aspect, we assume that the herd size slightly decreases with increasing age of the buildings, implemented by a correlation of -0.10 between herd size and construction year of the stables. All other correlations are assumed to be zero.[58]

8.2.4 LHS of a representative sample

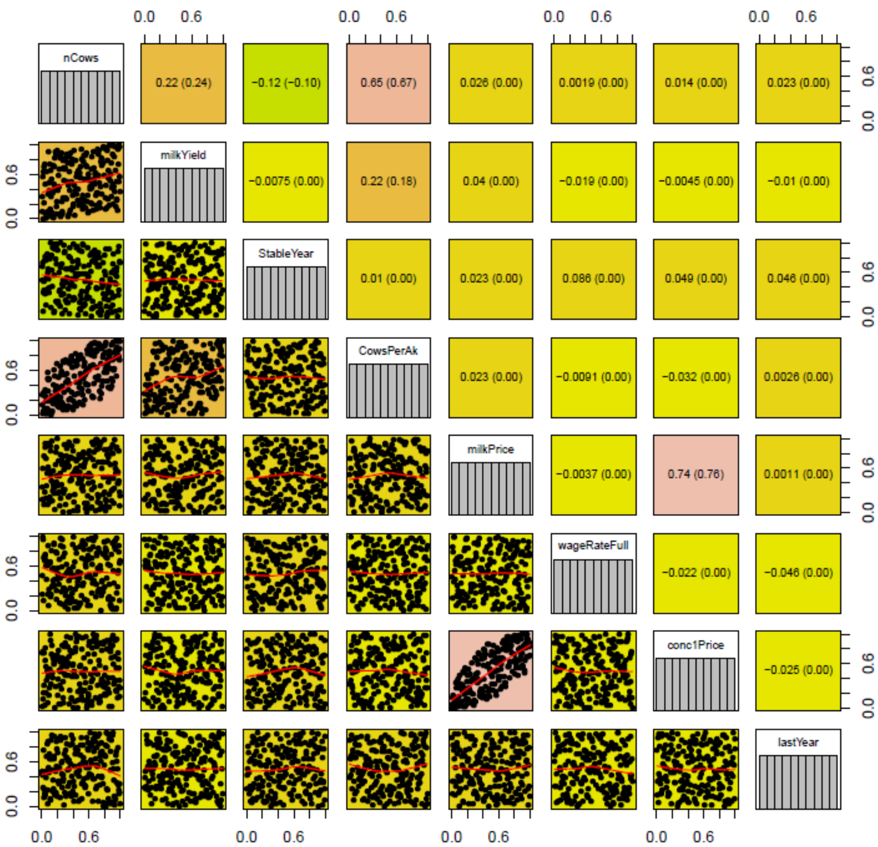
We used the known correlations and factor value ranges to determine a representative sample. To do so, we employed the statistical software R (version 2.15.1) for the DOE generation, specifically the LHS-package “lhs_0.10” in combination with the algorithm

[57] Pearson correlation coefficients with significance levels of ***=0.01, **=0.02, *=0.05

[58] With this assumption we can also ensure that the danger of multicollinearity in sampling outputs is diminished.

from IMAN and CONOVER (1982) to incorporate correlations. As the accuracy to which that algorithm can recover the correlation depends on the drawn sample, we performed the LHS a few thousand times for a sample size *n* of 200 and selected the LHS-sample with the best fit between the randomized and the given correlation matrix. The scatter plot matrix (figure 2) illustrates the random draws for each variable (normalized to the range [0, 1] expressing the value of the cumulative distribution function of each variable; cf. WYSS and JORGENSEN, 1998: p.7) depending on the other randomized variables. The single scatter plots of figure 2 illustrate that the sampling procedure outputs are smoothly distributed over the factor range for factors assumed to be not correlated such as in the case of *milkYield* against *StableYear*. The fit to the given correlations can be found in the upper right half of the matrix which shows the drawn (without parenthesis) and the given (in parenthesis) correlation factors. The improved LHS is hence efficient as it is space filling, shows a good fit to the given factor correlations (the mean percentage deviation of given and drawn correlations of the whole random sample is 7.76%) and markedly reduces the number of required input combinations to a manageable amount as required by CARRIQUIRY et al. (1998: p.507).

Figure 2: **Scatter plot matrix of 200 LHS draws for eight factors**



Source: own calculation and illustration.

The scatter plot matrix of the LHS (between 0 and 1) is translated to the actual level of each factor by assuming a uniform distribution function between the minimal and maximal values[59] reported in table 1. We excluded some implausible factor level combinations (e.g. very high milk yields in combination with very small herd sizes), leaving us with 155 single farm experiments. For each experiment, we simulated 20 different GHG emission ceilings with DAIRYDYN, covering a range up to 20% abatement requirement (between 80% and 100% of the baseline emissions) for each of the two indicators (actBased and refInd) to generate single farm MAC outputs.[60]

8.2.5 Statistical meta-modeling

In order to quantify which factors significantly impact MACs under the two indicators, a statistical meta-model is constructed for each indicator. These statistical response surfaces can be understood as "approximation of the simulation program's I/O transformation" (KLEIJNEN, 1999: p.116), in this study for MACs depending on farm attributes, GHG reduction levels and the indicator scheme applied.

This requires selecting an appropriate statistical estimator. As our non-standard LHS accounting for factor correlations is no longer orthogonal, we first tested for multicollinearity of the randomized sample variables (not critical, the variance inflation factors were below 10)[61]. A first graphical analysis of results revealed that in certain experiments, farms exit dairy production at more aggressive GHG emission ceilings. The exit decision is mainly driven by the opportunity cost of land (returns to land from the optimal farm program compared with land (rental) price) and labor (returns to labor at optimal farm program compared with off-farm wage), which depend in turn on farm characteristics and prices for outputs and inputs. Once a farm has exited, its GHG emissions will stay unchanged at zero and profits will no longer change. Hence, further tightening of the emission ceiling cannot deliver useful information on MACs. Leaving these zero observations in the sample would lead to biased results, while excluding them would omit information (censored data) and cause a sample selection bias[62] (HECKMAN,

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[59] Therefore, the following formula is applied: $F(x)*(b-a)+a=x$, where $F(x)$ is the LHS value of the factor (the value of the cumulative probability function), and *b* (upper bound) and *a* (lower bound) are the assumed boarders of the possible factor range (cf. table 1).

[60] For details concerning the formal derivation of MACs see LENGERS and BRITZ (2012).

[61] MARQUARDT (1970) suggests that serious collinearity is present for VIF-values above 10.

[62] TOBIN (1958) first showed that if censoring of the dependent variable is not considered in the regression analysis, an ordinary least squares (OLS) estimation will produce biased estimates.

1979; KENNEDY, 2008: p.265). Therefore, we apply a *two-stage Heckman estimation procedure* which in the selection equation first estimates the probability to exit farming (probit-model) and in the outcome equation (ordinary least squares (OLS) linear regression), conditioned on the first stage probability, the MACs. Accordingly, as standard in the Heckman approach, the inverse Mill's ratio calculated from the first stage probit-model was added as an explanatory variable to the second stage to correct for the self selection bias[63]. (HECKMAN, 1979; KENNEDY, 2008: pp.265-267) The estimation is performed with the R "SampleSelection" package (TOOMET and HENNINGSEN, 2008).

There is a second type of observations with zero MACs. LPs (linear programming models) might react with a basis change and thus non-smooth reactions to changes in binding constraints. This behavior is reinforced, as in DAIRYDYN, by the presence of integer variables. Due to a basis change, introducing a GHG emission ceiling might lead to a higher reduction than required. In our stepwise reduction simulations, a 1% increase in the enforcement level could lead to a reduction in GHGs of more than 2%. In this case, the next 1% reduction step will not require adjustments of the farm program. Thus, profits and abatement costs will not change in the next step for which, accordingly, the MACs become zero. These zero MAC observations are kept in the sample, but clearly will reduce the explained variance of the outcome equation. The latter, a multiple linear regression model, will react smoothly to changes in explanatory variables such as the GHG emission ceiling and cannot generate the kind of jumpy MAC curves which certain experiments might simulate.

8.2.6 Variable selection, transformations and interactions

There is no reason to assume that relations between MACs and farm attributes and further explanatory variables such as prices should be linear, or to exclude interaction effects in the design matrix. Hence, after some tests, we first also transformed variables to their square root and took the square of their reciprocal value. Additionally, we introduced interaction terms between all original variables. The resulting design matrix tends to be highly co-linear. Therefore, in a next step we estimated a linear regression between each explanatory variable and all others, dropping any explanatory variable with a multiple correlation above 94%. Afterwards, we used a backward selection strategy, dropping any variable with a significance level above 5% from the Heckman model.

[63] Inverse Mill's ratio is added so not to omit information of the explanatory variables of cases censored in the selection step. Explanation is also given in GIOVANOPOULOU et al. (2011: p.2177).

8.3 Results

The estimated coefficients of the meta-models are shown in the regression output tables in the appendix. The non-linear transformation of the variables and the presence of interaction terms render a direct interpretation of the estimated coefficients challenging. In order to visualize the effect of single factors on the MACs, we therefore separately varied the value of each factor over its range (cf. table 1), while fixing all remaining variables to their median. From there, we generated a set of new observations based on the transformations and interaction terms present in the estimated output equations (appendix 1 and 2). For this new set we simulated with the derived meta-models. This enabled the visualization of the course of MACs depending on level changes of the single significant regressors.

Effects on net on-farm MACs

The graphs of figure 3 indicate that MACs differ considerably between *GHG indicators*. Also, how they are affected by the different farm specific variables is crucially dependent on the indicator chosen. The right hand side figure shows that the simple actBased indicator provokes higher MACs in comparison with the more detailed indicator, which accounts for more flexible and low-cost abatement strategies (farm size reductions are the only options accounted by the actBased indicator as described in section 8.2.1).

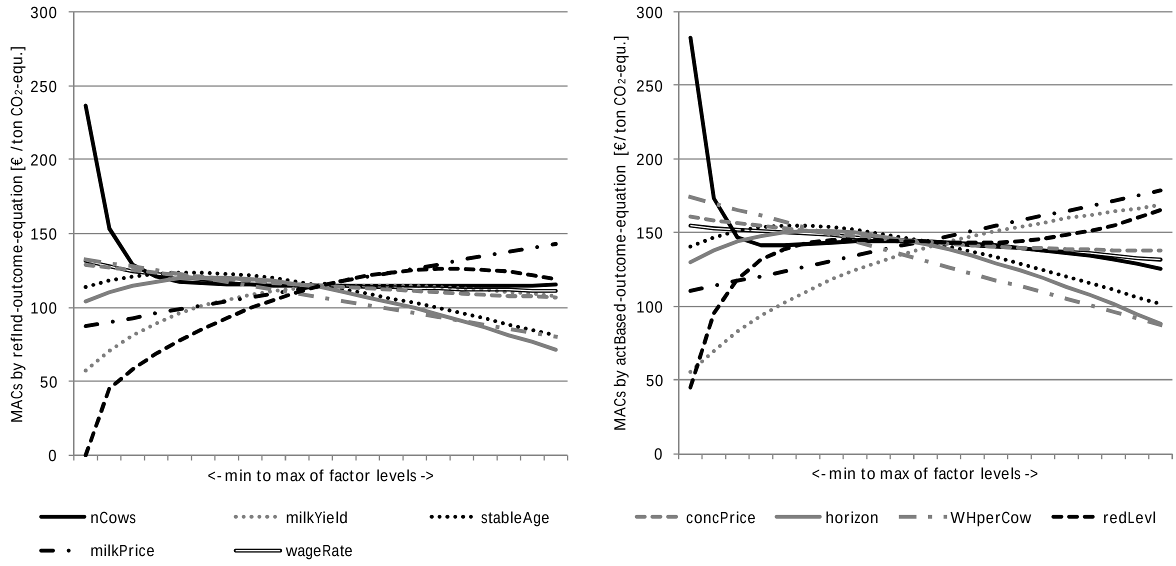
For all graphs, a steeper curve means a higher effect of factor level changes on the MACs over the factor range considered in the experiments. Hence, for both indicators the *emission ceiling* has the highest impact. As for the refInd estimates, MACs increase with diminishing rates if the mitigation level increases, the actBased indicator induces MAC curves which stay constant for intermediate reduction levels as the farm basically shrinks proportionally (reduction of cow numbers and acreages with constant losses of gross margins). For larger reductions above 15% under the simple indicator, the MACs increase as it becomes more likely in the sample to work half or full time off-farm, which leads to an increase in opportunity costs of labor. That finding shows the importance to reflect indivisibilities of labor in the integer approach underlying the model.

Differences between the indicators are mostly found for low reduction levels. Here, during the first 2 to 5% reduction of baseline emissions, the highest increase in MACs takes place, where MACs easily double between reduction steps. This shows that cheap abatement strategies are used first, with however only a limited reduction potential. At these moderate reduction levels, the more complex indicator benefits from offering low-cost abatement options not credited under the simple indicator such that its MACs are considerably below those induced by the simple indicator. For higher reduction levels MACs under both indicators reach a comparable level where abatement strategies under both indicators consist of farm size reductions. This triggers the observed steep increase in MACs once the abatement potential of low-cost measures is fully utilized and explains the decline in abatement cost advantages under the refInd with increasing abatement level.

The effect of the *milk price* shows that MACs are quite output price sensitive, independent of the indicator used. This can be easily understood from the fact that over a larger range of the GHG emission reductions, abatement efforts are linked to output reductions.

The effect induced by the *herd size* (nCows) of the farms is quite high for smaller farms below 40 cows. For larger farms, there seems to be no significant variation in MACs due to herd size variations. We interpret this finding by the fact that investment based mitigation options such as manure coverage, which allow for low-cost abatement, are not realized on small scale farms due to economics of scales, which however quickly level off.

Figure 3: **MACs in € ton^{-1} CO_2-equ. depending on factor levels under both indicators**



Source: own calculation and illustration.

Higher *milk yields* boost economic returns per cow and also per GHG emitted and thus provoke MAC increases under both indicators. The effect is less pronounced for the refInd (curve is less steep) for two reasons: first, the simple actBased indicator assigns a default emission factor to each cow, independent of the output level, whereas the refInd also recognizes the diminishing GHG emissions per kg of milk with increasing output level per cow and hence requires lower herd size reductions compared with the simple indicator. Second, as the refInd also accounts for improvements of the feed ration to lower emissions, stricter emission ceilings do not necessarily induce herd size reductions under the refInd, which lowers the impact of milk yield level variations on the MACs.

Independent of the GHG indicator chosen, variations of the *stable age* as well as the *simulation horizon* show nearly the same effect: they slightly reduce the MACs by increasing flexibility with respect to the distribution of emissions over time and to the timing of investment decisions and adherent path dependencies.

The *wage rate*, determining the opportunity costs for on-farm work, has a significant but moderately negative impact on the MACs as higher returns to off-farm work reduce the profit foregone from shrinking the farm operation.

Similarly, a lower labor productivity which increases the *working hours spend per cow and year* (WHperCow) reduces the MACs for both indicators (the effect is slightly smaller under the detailed refInd indicator). Reducing GHGs with a low labor productivity releases more labor for off-farm work, which dampens income losses compared with more labor efficient farms.

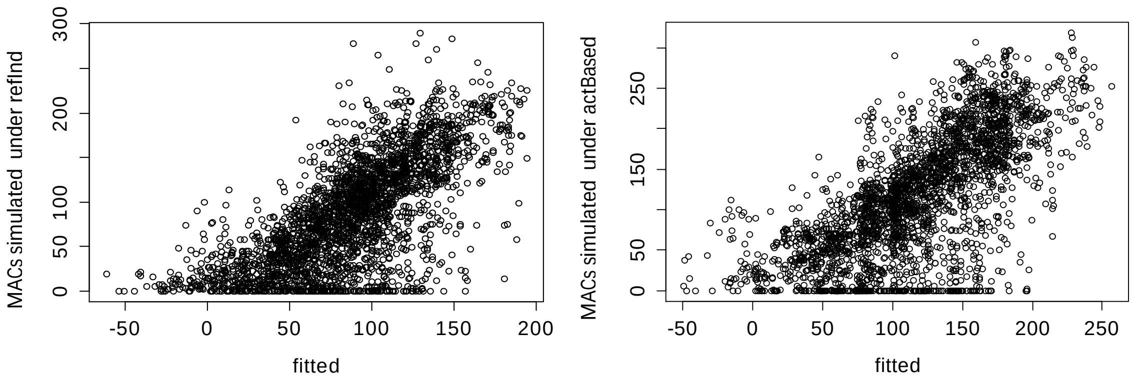
Higher *concentrate prices* let MACs to decline somewhat under both indicators, which is simply the opposite effect of an increased output price. The lower impact compared with the milk price reflects the cost share of concentrates.

8.4 Discussion of results

Figure 4 plots the fitted values of the meta-models against those simulated with the underlying complex bio-economic model. The reader should first note the thick cluster of zero MAC observations along the horizontal axis, which are rooted in basis changes in the MIP based bio-economic model. These observations, hardly meaningful for economic analyses, are smoothed out the by meta-models.

These zero observations contribute to the fact that we estimated a relatively low adjusted R^2 of about 0.46 for the actBased-meta-model and 0.47 for the refInd-meta-model. Generally, it is hard to achieve a good fit over the non-smooth simulation behavior of a LP or MIP with a regression model. The reader should also be aware that the fit is reported for relations between first differences (difference in profits in relation to difference in maximal emissions), and not for simulated variable profits. Against this background, the fit of the estimated meta-models can be valued as rather acceptable as it only consists of highly significant regressors with a significance level of at least 5%.

Figure 4: **Scatter plot of fitted values from meta-model against values simulated with bio-economic model**



Source: own calculation and illustration.

As with any type of statistical model, results are conditioned on the input data and hence only valid in front of the sampling background. Whereas the direction of impacts found for the factors can be motivated from economics theory and thus generalized, the best fit variable transformations and related parameter estimates might depend on the sampling design and are therefore probably only representative for dairy farm conditions in North-Rhine Westphalia, Germany, as reflected in the parameterization and structure of DAIRYDYN.

As simulation results were generated from specific assumptions about the length of the simulation horizon and the currently observed production techniques implemented in the model code of DAIRYDYN, an interesting question for further research is whether and how to reflect future technical progress in farming. Whereas it is obvious that innovations and their adaptation can change MACs (cf. AMIR et al., 2008), it is not clear how to account systematically for that effect in an economic simulation with a longer time horizon.

We want to remind the reader that single factor effects visualized in figure 3 are derived by fixing all other factors to their median. Clearly, the single factor MAC curves may change both with regard to their mean and slope when fixing the other factors to different levels. This may reorder the ranking of the factors with regard to their impacts on MACs. The selected curves are thus only illustrative of the statistical dependencies expressed by, and the simulations possible with the meta-models, while underlining the usefulness of the meta-models for systematic analyses.

8.5 Summary and conclusions

Our paper discusses the development of a statistical meta-model from simulations using the highly detailed bio-economic single farm optimization model DAIRYDYN, in order to analyze systematically key factors impacting marginal abatement costs for GHGs on dairy farms and to quantify their effects. DAIRYDYN was parameterized and the experiments set up to yield a set of results covering the relevant range of core attributes representative of the dairy farm population in North Rhine Westphalia, Germany. Specifically, we used a non-orthogonal Latin-Hypercube sampling approach, which accounts for correlations between factors for our design of experiments. As we observed for a non-negligible share of simulations farm exits which yield zero MACs for further emission enforcements, the meta-model was estimated based on a Heckman two-stage procedure to avoid selection bias. Our results deliver more farm level oriented analyses for the explanation of MACs compared with existing studies as mentioned in the introduction and hence give important insights into differences in MACs between farms.

We found the following main factors influencing the farm-level abatement costs on dairy farms, in order of importance: the GHG reduction target (in line with findings from KUIK et al., 2009: pp.1399-1400), size of the farm, milk price and milk yield level. Wage level, labor productivity, concentrate prices, simulation horizon and time of last investment in stables also impact the MACs, nevertheless at lower rates. We found that MACs increase quite strongly between a 2 and 5 % abatement level, a clear hint of a limited potential for low-cost abatement options in dairy farming. This conclusion is also reached by LENGERS et al. (2013a). Our findings thus suggest that MACs differ considerably between farms of different sizes and production intensities for the same GHG emission target, and react quite sensitively to changes in input and output prices. An interesting observation is the fact that MACs decrease if farms are allowed to distribute more flexibly the required GHG reduction over several years, a point also raised by FISCHER and MORGENSTERN (2005: p.2). Of equal importance, the GHG indicator employed has a strong impact on the level of induced MACs. All these observations are clearly relevant for policy discussion and design as they denote highly complex dependencies between the observed factors and farm-level MACs, which require more detailed analyses concerning heterogeneity aspects of MACs in the actual farm population.

To conclude, our study showed significant effects on the MACs both for farm attributes and prices, and for factors relating to policy implementation such as the GHG reduction target and the chosen GHG indicator. Therefore, we complement existing meta-model analyses, which relate MAC differences to the chosen methodological approaches only, while going beyond studies at the farm level, which delivered results for selected single farms.

An advantage of the derived meta-models compared with the application of the underlying complex simulation model is their easier integration into other modeling approaches and especially its faster execution time and lower storage needs (BRITZ and LEIP, 2009: p.267). Following the systematization of KARPLUS (1983) as well as ORAL and KETTANI (1993) the complex grey box model DAIRYDYN, which builds on detailed bio-economic causal relations, is transformed into a far simpler black box model (a set of statistically derived functions) where the logical and approximately also numerical relations are maintained. These meta-models can be used for analyses and explanation as we have done, but also to investigate future scenarios (KLEIJNEN, 1995: p.158) (no causal but logical dependencies represented by black box models). Hence, the meta-models (cf. appendix) derived are well suited for upscaling purposes to regional or sectoral level from single farm or farm group observations, while reflecting highly non-linear and complex relations between farm attributes and MACs, a point also underlined as important by SCHNEIDER and MCCARL (2006: p.285).

The underlying single farm model DAIRYDYN can be used to simulate any farm with a dairy production system matching its current structure and parameterization (no grazing in winter, loose housing systems, rather high mechanization level). Additionally, the sampling procedure we implemented makes possible an efficient and flexible adjustment of the sampling and simulation approach to derive GHG indicator dependent meta-models representative of other regions of interest by customizing the factor boundaries and assumed correlation terms. The approach can thus be used to come up with a collection of regional or national specific meta-modeling functions. In further studies these could be an important tool to analyze distributional aspects of GHG related environmental policies in the actual farm population.

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Appendix

Appendix 1: Output of the Heckman meta-model, Probit for farm exit and OLS for MACs under the refInd indicator ("factorname"_2 = square of the reciprocal value)

Tobit 2 (sample selection model)
2-step Heckman / heckit estimation
2858 observations (93 censored and 2765 observed)
45 free parameters (df = 2814)

Probit selection equation:

	Estimate	Std. Error	t-value	Pr(>|t|)	
(Intercept)	-0,083967	0,495614	-0,169	0,865	
nCows	0,020144	0,001782	11,306	< 2.00E-16	***
wageRate	-0,168394	0,029757	-5,659	1,68E-08	***
Horizon	0,106264	0,024862	4,274	1,98E-05	***
WHperCow	0,031782	0,005963	5,33	1,06E-07	***
redLevl	-0,074977	0,012081	-6,206	6,24E-10	***

Outcome equation:

	Estimate	Std. Error	t-value	Pr(>|t|)	
(Intercept)	5,79E+02	6,45E+01	8,968	< 2.00E-16	***
milkYield	-2,35E+00	8,19E-01	-2,871	0,004126	**
wageRate	-1,61E+01	4,94E+00	-3,252	0,00116	**
WHperCow	-2,60E+00	1,18E+00	-2,207	0,027415	*
redLevl	1,22E+01	2,08E+00	5,859	5,20E-09	***
nCows_2	7,87E+02	7,72E+01	10,194	< 2.00E-16	***
milkYield_2	-3,72E+03	3,35E+02	-11,094	< 2.00E-16	***
stableAge_2	-3,57E+01	4,88E+00	-7,329	3,01E-13	***
wageRate_2	1,06E+01	4,98E+00	2,121	0,034003	*
horizon_2	-1,41E+02	1,61E+01	-8,723	< 2.00E-16	***
redLevl_2	-1,23E-01	3,16E-02	-3,887	0,000104	***
nCows*stableAge	-2,20E-02	5,98E-03	-3,684	0,000234	***
nCows*milkPrice	2,90E-02	3,66E-03	7,91	3,67E-15	***
nCows*wageRate	-4,00E-02	7,19E-03	-5,566	2,85E-08	***
nCows*redLevl	-1,58E-02	3,39E-03	-4,664	3,24E-06	***
milkYield*stableAge	1,12E-01	1,93E-02	5,801	7,32E-09	***
milkYield*wageRate	1,21E-01	2,34E-02	5,176	2,43E-07	***
milkYield*concPrice	-1,13E-02	3,26E-03	-3,473	0,000523	***
milkYield*horizon	9,70E-02	1,85E-02	5,248	1,65E-07	***
milkYield*WHperCow	-1,43E-02	6,24E-03	-2,288	0,022192	*
stableAge*milkPrice	-2,80E-01	6,43E-02	-4,36	1,35E-05	***
stableAge*wageRate	5,22E-01	1,19E-01	4,371	1,28E-05	***
stableAge*horizon	-1,40E+00	1,09E-01	-12,86	< 2.00E-16	***
stableAge*WHperCow	1,80E-01	3,10E-02	5,792	7,72E-09	***
milkPrice*wageRate	6,73E-01	1,16E-01	5,818	6,62E-09	***
milkPrice*horizon	-3,53E-01	6,42E-02	-5,499	4,16E-08	***
milkPrice*redLevl	2,60E-01	4,64E-02	5,604	2,30E-08	***
wageRate*concPrice	-7,33E-02	2,34E-02	-3,135	0,001738	**
wageRate*horizon	3,99E-01	1,17E-01	3,408	0,000664	***
wageRate*WHperCow	-1,40E-01	4,29E-02	-3,273	0,001077	**
concPrice*concPrice	2,46E-03	1,05E-03	2,339	0,019399	*
concPrice*WHperCow	9,94E-03	4,50E-03	2,209	0,027235	*
horizon*WHperCow	6,85E-02	2,90E-02	2,362	0,018253	*
horizon*redLevl	-1,58E-01	5,16E-02	-3,063	0,002212	**
WHperCow*redLevl	-9,62E-02	1,60E-02	-6,023	1,94E-09	***
redLevl*redLevl	-3,44E-01	3,25E-02	-10,586	< 2.00E-16	***

Multiple R-Squared: 0.4792; Adjusted R-Squared: 0.4724
Error terms:

	Estimate	Std. Error	t-value	Pr(>|t|)	
invMillsRatio	-89,22	13,523	-6,598	4,97E-11	***
Sigma	49,731	NA	NA	NA	
Rho	-1,794	NA	NA	NA	

Signif. Codes: 0´***´ 0.001´**´ 0.01´*´ 0.05´.´

Appendix 2: Output of the Heckman meta-model, Probit for farm exit and OLS for MACs under the actBased indicator ("factor name"_2 = square of the reciprocal value, "factor name"_sqrt = square root)

Tobit 2 model (sample selection model)
2-step Heckman / heckit estimation
2857 observations (87 censored and 2770 observed)
43 free parameters (df=2815)

Probit selection equation:

	Estimate	Std. Error	t-value	Pr(>|t|)	
(Intercept)	-1,707229	1,000509	-1,706	0,088051	.
nCows	0,026671	0,002778	9,602	< 2.00E-16	***
milkYield	-0,012157	0,004615	-2,634	0,008487	**
milkPrice	0,073575	0,025169	2,923	0,003491	**
wageRate	-0,214934	0,034862	-6,165	8,04E-10	***
Horizon	0,105747	0,02755	3,838	0,000127	***
WHperCow	0,040794	0,006776	6,021	1,96E-09	***
redLevl	-0,068917	0,012724	-5,416	6,60E-08	***

Outcome equation:

	Estimate	Std. Error	t-value	Pr(>|t|)	
(Intercept)	3,87E+02	5,31E+01	7,28	4,30E-13	***
nCows	7,95E-01	3,14E-01	2,531	0,011433	*
wageRate	-2,52E+01	6,35E+00	-3,969	7,40E-05	***
redLevl	-4,97E+01	5,41E+00	-9,183	< 2.00E-16	***
nCows_2	1,12E+03	1,23E+02	9,132	< 2.00E-16	***
milkYield_2	-3,23E+03	4,53E+02	-7,119	1,38E-12	***
stableAge_2	-4,63E+01	6,29E+00	-7,352	2,55E-13	***
horizon_2	-1,82E+02	2,01E+01	-9,038	< 2.00E-16	***
redLevl_sqrt	1,82E+02	1,78E+01	10,22	< 2.00E-16	***
nCows*nCows	-1,40E-03	5,46E-04	-2,562	0,010459	*
nCows*stableAge	-3,38E-02	7,77E-03	-4,355	1,38E-05	***
nCows*milkPrice	2,59E-02	6,04E-03	4,285	1,89E-05	***
nCows*wageRate	-3,88E-02	1,05E-02	-3,679	0,000238	***
nCows*horizon	-2,38E-02	7,53E-03	-3,166	0,00156	**
milkYield*stableAge	9,99E-02	2,16E-02	4,632	3,79E-06	***
milkYield*wageRate	1,44E-01	2,85E-02	5,061	4,45E-07	***
milkYield*concPrice	-1,24E-02	2,43E-03	-5,103	3,56E-07	***
milkYield*horizon	6,44E-02	2,30E-02	2,803	0,005098	**
milkYield*WHperCow	-2,41E-02	7,14E-03	-3,383	0,000727	***
stableAge*wageRate	6,64E-01	1,55E-01	4,295	1,80E-05	***
stableAge*concPrice	-4,40E-02	1,32E-02	-3,337	0,000857	***
stableAge*horizon	-1,68E+00	1,29E-01	-12,986	< 2.00E-16	***
stableAge*WHperCow	2,41E-01	3,65E-02	6,609	4,63E-11	***
milkPrice*wageRate	7,30E-01	1,40E-01	5,202	2,11E-07	***
milkPrice*horizon	-1,65E-01	7,72E-02	-2,14	0,032453	*
milkPrice*WHperCow	-5,68E-02	2,17E-02	-2,621	0,00882	**
wageRate*concPrice	-8,62E-02	2,73E-02	-3,154	0,001628	**
wageRate*horizon	4,95E-01	1,49E-01	3,324	0,0009	***
wageRate*WHperCow	-1,05E-01	4,99E-02	-2,096	0,036215	*
concPrice*concPrice	4,73E-03	1,08E-03	4,39	1,17E-05	***
concPrice*redLevl	2,75E-02	9,76E-03	2,823	0,004795	**
redLevl*redLevl	7,47E-01	9,96E-02	7,501	8,46E-14	***

Multiple R-Squared: 0.4715; Adjusted R-Squared: 0.4653
Error terms:

	Estimate	Std. Error	t-value	Pr(>|t|)	
invMillsRatio	-126,803	11,743	-10,8	< 2.00E-16	***
Sigma	62,692	NA	NA	NA	
Rho	-2,023	NA	NA	NA	

Signif. Codes: 0´***´ 0.001´**´ 0.01´*´ 0.05´.´

Appendix 3: Marginal effects of the selection equation for exit decisions under the refInd indicator

Marginal effects of first stage

"[1] ""probitmxfull"""

	mfx	SE	Mean Value	z	Pr(>|z|)
nCows	8,30E-05	7,35E-06	133,2588	11,30615	1,22E-29
wageRateFull	-6,94E-04	1,23E-04	10,5398	-5,658936	1,52E-08
Horizon	4,38E-04	1,02E-04	14,76351	4,274118	1,92E-05
WHperCow	1,31E-04	2,46E-05	44,60066	5,329878	9,83E-08
redLevl	-3,09E-04	4,98E-05	10,50872	-6,205917	5,44E-10

Appendix 4: Marginal effects of the selection equation for exit decisions under the actBased indicator

Marginal effects of first stage

"[1] ""probitmxfull"""

	mfx	SE	Mean Value	z	Pr(>|z|)
nCows	2,64E-05	2,75E-06	133,18825	9,601836	7,85E-22
milkYield	-1,20E-05	4,57E-06	83,11021	-2,633916	8,44E-03
milkPrice	7,28E-05	2,49E-05	30,41257	2,923294	3,46E-03
wageRateFull	-2,13E-04	3,45E-05	10,53858	-6,165292	7,04E-10
Horizon	1,05E-04	2,73E-05	14,76343	3,838432	1,24E-04
WHperCow	4,04E-05	6,71E-06	44,60428	6,020696	1,74E-09
redLevl	-6,82E-05	1,26E-05	10,50857	-5,416269	6,09E-08

Chapter 9: **Conclusions and outlook**

The overall objective of this work was to investigate the dependencies between GHG indicators for the crediting of GHG emissions to dairy farms and abatement costs (profits foregone) for GHG mitigation on single farms. This is important because market based instruments regulating GHG emissions like tradable permits or a taxation system require the use of such an indicator and will be impacted in their performance by the construction of the chosen GHG calculation scheme. Additional aspects relating to the indicator construction like GHG accounting accuracy and feasibility of the calculation schemes were analyzed. This was done not only to get deeper knowledge about the ability to trigger low-cost abatement, but also to gain insights into administrative burdens and accuracy aspects that are essential for any discussion about potential contribution of dairy production to national GHG reduction efforts.

In order to pursue the overall objective, some methodological and technical approaches were developed during the project work (*relating to the three methodological working objectives which were defined in the introduction of this dissertation* (see p.4)). The bottom-up approach DAIRYDYN was developed to simulate single dairy farms over time (chapter 5). Additionally, a set of differently detailed GHG indicators was defined, linked to the different modules of DAIRYDYN to derive indicator depending single-farm costs for the abatement of GHGs. For a statistical analysis of single factor effects (farm characteristics, prices, wages, reduction level, indicator) on the resulting MACs, an automated sampling and simulation procedure was developed to generate simulation outputs for a systematic meta-modeling analysis (chapter 8). These different tools were then used to generate scientific results on the overall objective. These results are presented in chapters 5 to 8.

Conclusions regarding the indicator effect on the potentials of cost efficient abatement, accuracy, and feasibility aspects:

The results obtained highlight the strong dependencies between GHG indicator construction and feasibility, accuracy, and the ability to trigger low-cost abatement. MACs are significantly impacted by the construction of the GHG quantification scheme, with lower MACs under detailed indicators. Also the GHG accounting accuracy increases with the complexity of the indicator’s definition. This directly impacts the bias between accounted MACs at the farm level and the actual MACs under real (and not the accounted) reduction amounts (cf. chapter 7). Logically, this bias increases dramatically with decreasing accuracy in emission accounting. Contrary to that, highly detailed indicators are less feasible due to availability and reliability of the necessary farm-level data. The meta-analysis in chapter 8 illustrates that the statistical dependencies between MACs and farm attributes, prices, emission targets and indicator construction show highly complex structures.

In terms of economic efficiency, to date abatement measures in dairy farming could contribute little to reduction efforts as - irrespectively of the indicator chosen - abatement costs on dairy farms are high. As long as no high rates of technical progress (which would potentially lower MACs) are to be expected in agricultural production these low-cost abatement measures can only be addressed with highly detailed GHG calculation schemes (cf. section 7.6; p.147). Once the small share of agriculture or dairy production in the overall GHG inventories (about 5.6% in German agriculture, see chapter 2) is taken into account, the actual mitigation effect of cost efficient measures would be negligible. So, it is very questionable that these small potentials would legitimate high administrative burdens and increased transaction costs, as transaction costs may annihilate even these rather limited abatement potentials. On the other hand, easily feasible simple indicators would increase MACs dramatically for low abatement levels as low-cost measures are not accounted for. They would therefore not lead to any abatement in dairy production under a price-based regulation system that directly (emission trading) or indirectly (taxes in the agricultural sector) considers relative marginal abatement costs of different sectors.

Furthermore, the rather high accounting bias of the simple indicators (chapter 7) reduces the economic efficiency of market-based instruments due to the difference between on-farm and societal MACs. Any market-based instrument thus requires highly detailed accounting schemes to guarantee economic efficiency, a fair treatment of agents (see chapter 7) as well as a reduction of adverse political decisions (KESICKI and STRACHAN, 2011). Additionally, beside heterogeneous structures in MAC curves that are to be expected for the actual farm population, MACs may also bear high uncertainties due to price sensitivities (chapter 8). And with regard to the potential bias between accounted and actual emissions and relating distortions of assumed cost implications, this again may contort the targeted precision of any market-based GHG reduction policy.[64] These uncertainties will finally impair the predictability of economic and environmental effects. Hence, in an agricultural context it seems hardly feasible to construct appropriate indicators that would lead to unbiased results with acceptable administrative burdens.

It can thus be concluded that for the time being, the inclusion of dairy production into GHG reducing price-based policy regimes (tradable permits or taxation systems) is not an appropriate way to efficiently activate GHG abatements in this sector (see chapter 7) which is consistent with conclusions of ANVEC (2011a, 2011b).

A feasible strategy could be the application of statutory requirements to directly activate appropriate low-cost measures. Statutory requirements may be easy to control via existing building laws or requirements of the EU common agricultural policy (nitrate directive, etc.). Furthermore, this may cause synergy effects with other externalities of agricultural production (e.g. in case of manure coverage techniques, undesired induction of water as well as ammonia emissions can be reduced). For the definition of efficient statutory requirements, more analyses have to be carried out to systematically indentify those measures that lead to low-cost abatement for the majority of farms. In line with conclusions of VELLINGA et al. (2011: p.194), this could be the most feasible and preferable approach from a political as well as a farm level perspective, as farmers prefer simple options in facing implemented GHG ceilings. Following ANVEC's suggestions, "[...] alternative coverage policies of voluntary opt-in and emission reduction offsetting credits[65] might [...]" also be an advantageous approach. These incentive schemes may animate agricultural emitters that are able for low-cost abatement to realize these cost efficient measures on their farms. These opportunities may activate low-cost abatement potentials without the expensive inclusion of the whole sector to market based regulation systems. (ANVEC, 2011a)

Concluding remarks on the methodological approaches that have been developed:

The construction of the single farm model DAIRYDYN (explained in detail in chapter 5) and the obtained results illustrate that a highly detailed bio-economic resolution

[64] Although market based instruments theoretically lead to macro-economically efficient abatement, in the agricultural context this is not the case as GHG indicators are required (not being able to directly measure the actual emissions) and MACs may be highly biased depending on the accounting accuracy of the applied indicator (indicated by the differences between on-farm and societal MACs in chapter 7).

[65] These credits are coupled to specific investments for GHG reduction offsetting measures.

of a model approach is a prerequisite to analyze all facets and influences of the GHG indicator choice in environmental policy design. The supply side model enables accounting for trade-offs and interaction effects between different mitigation options and gases, an important aspect often neglected in existing studies. Of special importance seems the dynamic and mixed integer character of the optimization model, as well as the optimization over a longer time span. The latter allows for investment-based mitigation strategies which show significant impacts on MACs. The dynamic setting also aids to incorporate the effect that single measures implemented in one year may influence the costs of other measures in subsequent periods.[66] This point is also seen as necessary by KESICKI and STRACHAN (2011: pp.1199-1202). DAIRYDYN hence serves as a powerful tool, enabling highly detailed single farm simulations for the work on environmental as well as economic questions.

The five IPCC-based GHG indicator schemes explained in chapter 4 show clear differences in the level of detail in required on-farm process information. They are based on scientifically consistent methodologies to ensure that they systematically take into account the predetermined system boundary (farm gate). This is of special importance for the scope of analyses carried out, avoiding problems like double counting of emissions. The calculation schemes of the GHG indicators indicate the differences between default and highly detailed GHG indicators, which is essential for the analysis presented in chapter 7 and 8 with regard to accuracy, feasibility and abatement cost aspects.

The sampling and meta-modeling approach (chapter 8) was developed in order to get deeper insights into the dependency between abatement costs and indicator schemes, farm attributes and other aspects influencing the income level of a farm (e.g. in- and output prices). This approach complements existing meta-model analyses, which relates MAC differences to the chosen methodological approaches[67] only, while going beyond studies at the farm level, which can only deliver results for selected farms. The analysis highlighted that the sampling design is highly efficient to define a set of representative farm experiments (that is: specific parameterization of farm characteristics serving as single-farm definition for simulations with DAIRYDYN; cf. chapter 8.2) for a specific farm population. Thereby it is easily adjustable to the distribution of single farm characteristics

[66] For example, investments in manure coverage lower emissions from storage but will potentially increase emissions from fertilization and hence drive the costs for GHG abatement by different types of manure application.

[67] E.g. supply side models, engineering models or equilibrium models.

and prices in the investigated farm population. Further, the results of the experiments carried out by DAIRYDYN can be used for statistical meta-modeling. This offers a highly challenging aspect as the results obtained by DAIRYDYN and the definition of the single experiments can give microeconomic information of a large set of single dairy farms. These can then be used for systematic statistical analyses, for example informing on the most relevant characteristics for upscaling of single farm results. Implementing an automated selection procedure, only significant factors and their quantitative effect on the MACs under different GHG indicators remain in the resulting MAC functions. Thereby they reflect highly non-linear and complex relations between farm attributes, prices and MACs on the micro-level. Therefore, they are well suited for the implementation into other model approaches that pursue more macroeconomic objectives and that are less detailed in single farm process description.

Research outlook:

The presented work offers some hints towards additional areas of interest concerning improvements of the methodological approaches as well as further research questions.

Most aspects concerning the methodological approaches of this study that have to be developed further are related to the underlying model DAIRYDYN. This powerful tool could be improved to make further in-depth investigations on the above stated objectives or to expand the investigated agricultural production processes. To date, the model is only appropriate to build up specialized dairy farms with slatted floor stable systems. A more diversified model approach may lead to new insights in the topic of indicator-relating GHG mitigation costs and the overall ability of agriculture for a low-cost GHG abatement. Diversified farms with additional features (e.g. biogas production, straw based and slatted floor stable systems, alternative tillage procedures, other livestock categories, recognition of other gas types or environmental effects[68]) may offer new insights into chosen abatement strategies under emission regulation.

As MACs are highly sensitive to farm characteristics an expansion of the investigation to the sectoral level in Germany is demanded to investigate MAC distributions of the whole population. Additionally, considering the background of

[68] Also CO_2 for sequestration and release by change of tillage techniques or land use change, ammonia, phosphor, odor, etc. For this study CO_2 accounting for sequestration and release by changes of tillage techniques was neglected as changes in tillage practices were not captured by DAIRYDYN. But in the meantime the model has been enlarged by different tillage techniques, which makes also a CO_2 accounting by the indicator schemes reasonable.

increased price volatilities on agricultural factor markets during the last years, price volatilities may constitute an increase in uncertainty for MACs relying on price sensitivities of abatement measures. And as the choice of indicator also impacts the degree of single factor effects (of prices, wages, farm characteristics…; see chapter 8), uncertainty aspects in MACs consequently also strongly depend on the GHG indicator scheme which makes the overall discussion more complicated. This point is also emphasized as a challenging aspect by KESICKI and STRACHAN (2011: p.1202) as well as VERMONT and DECARA (2010: p.10). These uncertainties in MACs will impact the potential compliance or precision of the targeted reduction policy on a private and especially on a societal level.

Hence, further research in this field is required to get deeper insights into the MAC distribution of the whole sector, probability distributions of MACs on single farms as well as to quantify cost uncertainties of single abatement measures that rely on price sensitivities. This could give additional insights in this highly complex discussion about potentials and practicability of GHG reductions in the dairy sector in particular or the agricultural sector as a whole.

Furthermore, deeper investigations have to be done with regard to sectoral effects, including market feedbacks, adjustment of production amounts and price changes, aspects also discussed by PÉREZ (2006: pp.170-172) and PÉREZ et al. (2009: p.305). In this regard, it is a challenging task for future research to raise the restricted single farm investigations of this study to a higher aggregation level. To this end, the meta-modeling approach may be a highly valuable tool. It can be adjusted for the preparation of regional indicator dependent MAC functions. These can serve as input for more aggregated modeling approaches (like equilibrium models) that are able to implement also competition on factor markets and price feedbacks.

To sum up, estimated results clearly show that the understanding of MACs for GHG abatement is a highly complex issue which needs deeper insights and analyses with regard to the farm population to be included into mitigation efforts. Thereby, it is also important to investigate potentials and practicability aspects of accounting and regulation schemes. Otherwise, as also concluded by KESICKI and STRACHAN (2011), political decisions under usage of unreflected MAC curve estimates can lead to biased results. The results of this study clarify the importance of a discussion about the construction of any indicator based environmental policy design in the agricultural context. In this regard the policy instrument, the targeted abatement level and the cost efficiency are not to be separated from discussions on the externality-quantifying indicator's definition, a point mostly ignored by scientific studies up to now.

For this purpose, this study builds a promising starting point for further analyses as the methodological approaches developed (dynamic MIP-model DAIRYDYN, sampling algorithm and meta-modeling procedure) and initial results obtained offer potentials and directions for additional research activities in this challenging field.

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