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Trade-off of feasibility against accuracy and cost efficiency in choosing indicators for the abatement of GHG-emissions in dairy farming

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Trade-off of feasibility against accuracy and cost efficiency in choosing indicators for the abatement of GHG-emissions in dairy farming

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Abstract

There is a broad discussion about the inclusion of agriculture into greenhouse gas reduction efforts, such as the Kyoto mechanism. As most agricultural GHG emissions stem 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 analysed trade-offs between these indicators´ feasibility, measurement accuracy and abatement costs based on emission reduction simulations with a highly detailed single farm optimisation model. Results indicate that the trade-offs depend both on farm characteristics and on the targeted reduction level. In particular, advantages of detailed indicators decrease for higher abatement levels.

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

JEL classification: Q12, Q15, Q18

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1 Introduction

With the Kyoto Protocol industrial nations agreed to reduce greenhouse gas (GHG) emissions until 2012 by about 5.2% relative to 1990. In the so-called burden-sharing agreements, country-specific goals were assigned to individual EU members. (UNFCCC 2009) New reduction goals are planned to be enacted on the 2012 UN climate conference in Katar. To pursue these goals, agriculture, emitting an estimated 10 to 12% of yearly global GHGs (Niggli 2009, p.1), probably has to contribute in the future to national and global emission reduction aims. Policy instruments such as tradable emission permits, emission taxes or statutory requirements, which are already implemented in industrial sectors, might thus be expanded to the agricultural sector. That requires the monitoring of agricultural emissions, in the best case by relying on actual measurements. However, in the case of non-point sources such as diffuse gaseous emissions of methane (CH_4), nitrous oxide (N_2O) and carbon dioxide (CO_2) from agricultural production systems, a direct measurement is not practicable (Osterburg 2004, p.209). Emissions have to be calculated instead, drawing on observable attributes of the investigation unit (e.g. animal number, crop acreages, fertiliser use, fodder ingredients). This study will call an accounting methodology for GHG emissions a GHG indicator.

Most of the existing studies on accounting and mitigation of GHGs from agriculture provide results for one specific indicator only, in most cases based on IPCC formulas or default values (e.g. DeCara et al. 2005; Lesschen et al. 2011; Olesen et al. 2006, p.209). Contrary to that, based on a bio-dynamic model approach for dairy farms named DAIRYDYN, Lengers and Britz (2012) highlighted the fact that GHG indicators influence abatement strategies and costs. They analysed indicators differing in complexity and level of aggregation and found that emission reduction strategies and the related profit losses depend to a large extent on the GHG indicator chosen.

We build on their work to analyse in more detail the effect of the different indicator schemes on the accuracy of GHG calculation from dairy production systems, the induced mitigation strategies and related abatement costs. Furthermore, we relate these results to differences in reduction requirements and different farm characteristics. Therewith, we want to contribute to a better understanding on the impacts of indicator construction on abatement costs. This is important for the discussion about political relevant indicator schemes (Walz et al. 1995) as an informed decision requires an in-depth understanding of the compromises made when increasing the feasibility of indicators at the detriment of accuracy and abatement cost efficiency, a point also stated by Osterburg (2004, p.214).

When facing policy instruments related to GHG reduction, entrepreneurs will change production processes or adopt new abatement strategies according to the effect their measures have on the specific GHG calculation scheme in place. Specifically, only abatement options will be recognised that are accounted for by the applied indicator scheme. Implementing an indicator which covers more abatement measures enlarges the decision maker's set of abatement options and thus might allow him to adopt a more cost-efficient abatement strategy.

Furthermore, as GHG indicators provide only a proxy of actual emissions, indicators may over- or underestimate the actually emitted or reduced emissions. This induces a difference in abatement cost from an *in-plant* perspective – relating profit losses to accounted emissions – and *actual* abatement cost based on profit losses and actual emission reduction. We will call these two types of abatement costs *net-private* and *net-social* costs. *Net-private* abatement costs relate the costs induced on a farm level by the application of abatement strategies to the accounted emission reduction, whereas *net-social* abatement costs relate the costs of farm level abatement to the actual abated emission amounts. In order to calculate "actual emission reductions" we design a reference indicator that accounts for all emission changes in our model. We term the costs calculated by our model as "net-"costs because we exclude measurement, administration and control costs for the emission quantification.

The normalisation of *net-private* abatement costs to *net-social* abatement costs is important to make indicators comparable because each of them is based on its own emission quantification procedure, typically resulting in different GHG estimates for an identical production plan[1]. Typically, as shown later, there is a strong correlation between accuracy and the detail in covered abatement options. If the choice of indicator would only impact abatement costs, one would hence select the ones covering more abatement options. But such detailed indicators typically require data about farm processes which are costly to monitor and/or control. In addition, some of these data might be needed to calculate abatement options, which only lead to minor reduction in abatement costs, contribute little to improved accuracy or which even are not realised on a specific farm. The main aim of the paper is to enquire for the different GHG indicators trade-offs between accuracy, abatement costs and feasibility of monitoring and controlling the required data.

The remaining paper is organised as follows: the next section offers a literature review on existing studies applying specific GHG accounting schemes from which requirements of suitable indicators are derived. Afterwards we describe the bio-economic model DAIRYDYN and its application in our context. Using differently detailed farm level data we define different indicator schemes drawing on IPCC (2006) guidelines, where necessary updated to reflect newer research findings. In a later subsection these indicators are related to abatement options covered in the model. The result section highlights differences in the accuracy of estimated GHGs and in GHG abatement costs between indicator schemes for a set of differently specified dairy farms. We analyse the influence of differences both in GHG reduction requirements and farm attributes - herd size and milk yields - on accuracy of estimating GHGs and abatement costs of different indicators. This also facilitates a brief discussion with respect to the practical applicability of these

[1] Compare to Lengers and Britz (2012).

indicators, both from the viewpoint of policy implementation and of farmers facing GHG policy instruments. Finally, we summarise and conclude.

2 Literature

Indicators have been used extensively in different industry sectors (Ridgley 1996), not least in the context of policies related to the mitigation and adaption of GHGs. There is now a vivid discussion regarding if and how agriculture could be included in GHG abatement efforts. A requirement for any such step is an indicator system as a technological control parameter for GHG emissions on a farm level (Osterburg 2004, pp.211; Scheele et al. 1993, p.298). Research work in that field uses indicators to calculate emissions from agricultural activities and derives in some cases also mitigation-related marginal abatement costs (MACs) (e.g. Breen 2008; DeCara et al. 2005; Hediger 2006; Olesen and Schelde 2008; Perez 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. Emission calculations in the reviewed studies were overwhelmingly based on IPCC equations or default values (IPCC 2006 or earlier versions). Only a few developed their estimations from experimental emission measurements on farms (e.g. Jungbluth et al. 2001; Ngwabie et al. 2009).

The degree of detail regarding GHG emission calculations in the different studies varies greatly, depending on data available and the research question. Breen (2008) based his calculations only 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 per animal and as a function of milk yield. In a study estimating GHG emissions from agriculture for Baden-Württemberg, a state of Germany, Neufeld et al. (2006, p.239) found that, with an R^2 of 0.85 and $p<0.01$, stocking rate seemed to be a good 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., based on raw nutrient intake (1993) as well as milk yield, body weight and type of roughage (1995), to discuss reduction potentials of abatement options in animal husbandry. Conclusions about N_2O release from excreta are drawn following N-excretion functions of 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 feed gross energy intake to calculate methane emissions from ruminants plus an equation from Bouwman (1989) for N_2O quantification, which solely bases on total N fertiliser doses.

Available studies have thus used quite different indicators with regard to the level of detail and the aggregation of relevant input variables. However, as each study only uses one indicator, they are unable to analyse how differently designed accounting methods impact their emission estimates and abatement costs if GHG reductions are implemented.

To our knowledge, Durandeau et al. (2010) firstly examined the influence of different detailed emission accounting schemes on marginal abatement costs (MACs) for N_2O from agricultural processes. They concluded that for an 8% emission reduction, induced MACs with a second-best indicator were about 7 times higher than with a first-best scheme, which incorporates a more detailed emission accounting to process variables. The impact of the GHG indicator construction on the possibilities of cost-efficient mitigation was also shown by Lengers and Britz (2012) offering illustrative simulation results of GHG abatement costs on dairy farms for differently designed GHG calculation schemes. Besides the differences in *net-private*[2] abatement cost efficiency between indicator schemes, they also pointed out that the measurement accuracy directly impacts its difference to the *net-social*[3] cost efficiency.

To conclude, different sets of indicators can be found in literature, but to our knowledge there is no systematic comparison of different possible indicators yet.

Criteria for appropriate indicators can be derived from findings by 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). They can be summarised to three criteria: *feasibility*, *accuracy* and *cost efficiency*. *Feasibility* refers to the monitoring and control of the necessary data required at farm level. Hence, it depends on the existence or potential to develop new reporting systems on farm level feeding an indicator with the necessary data. Feasibility generally diminishes with increasing data requirements. *Accuracy* is linked to the ability of emission indicators to approximate actual emissions and relates to consistency and detail of calculation schemes (Schröder et al. 2004, p.20). *Cost efficiency* is in our study discussed from a private and societal perspective: (1) net-private cost: the on-farm abatement costs provoked by an indicator, depending on the cost efficiency of the abatement measures covered by the indicators, they relate to accounted GHG emissions; and (2) net-social costs: the on-farm abatement costs related to actual GHG abatement which hence also depend on the relation between accounted and actually emitted GHGs, and thus relate also to accuracy.

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[2] Notation comparable to the not-normalised abatement costs in the study of Lengers and Britz (2012).

[3] Notation is equivalent to the normalised abatement costs by Lengers and Britz (2012).

An adequate design of a GHG indicator scheme typically requires a compromise because of trade-offs between these three requirements (Walz et al. 1995).

3 Model and GHG indicators

3.1 *The Model*

The model DAIRYDYN, applied in this paper, is a fully dynamic mixed integer linear optimization model for the simulation of detailed dairy farm production and investment plans over several years, possibly constrained by GHG emission ceilings (Lengers and Britz 2012). That bio-economic model approach, covering in high detail farm processes which impact GHG emissions as decision variables, maximises the net present value of future profits over different states of nature and enables the implementation of different emission accounting schemes. As the decision maker of each farm is assumed to be fully informed and rational, simulation results reflect best-practice behaviour. The mixed integer character of the model allows the implementation of investment-based long-term abatement strategies as binary decision variables and the reflection of indivisibilities in labour-use decisions.

GHG emissions in the model refer to the whole-farm and thus are aggregated from different sources (animal, soil, manure, etc.). As a supply side model with an explicit description of the production feasibility set, it eases the calculation of abatement costs (Vermont and DeCara 2010). It simulates a single farm at given prices, in contrast to partial equilibrium models which typically work on a more aggregated scale, but endogenise price changes by incorporating market feedbacks. Compared to so-called engineering models our approach facilitates a detailed technological description on farm level and additionally also endogenously calculates interactions between different GHG mitigation strategies, which are important for consistency (MacLeod et al. 2010, p.200).

By coupling modules covering on farm processes such as animal husbandry, feeding, manure and mineral fertiliser handling or land use, bio-physical interactions are depicted, including possible trade-offs between emissions of different gases from identical or different sources (Weyant et al. 2006). This is based on detailed response and mass flow functions within and between modules. This process-based approach accounts more accurately for interactions between bio-physical processes compared to e.g. MacLeod et al. (2010) who accounted for only two abatement measures at a time linked by fix interaction factors. In DAIRYDYN, abatement strategies consist of profit maximal simultaneous changes of decision variables in different modules which will have markedly different effects compared to the independent appraisal of mitigation (shown by Schneider and McCarl 2006 using the ASMGHG-model). This should allow for a more realistic framework to analyse farm optimisation decisions, for example when GHG ceilings are implemented.

A single run is based on the definition of basic farm characteristics such as the starting herd size, the milk yield potential of the cows, age of stables as well as prices and costs. Other parameters, such as the labour, stable and land endowment, can either be selected manually or derived based on a set of engineering rules.

In order to derive abatement costs, total farm emissions can be stepwise reduced by choosing the number of reduction steps and the percentage amount of baseline emissions (baseline emissions are the GHGs emitted by the farm without any emission restriction) to be reduced in each step.

The total "net-private" GHG abatement costs (ACs) for a single farm are defined by the simulated profit loss against the baseline provoked by the implementation of an emission ceiling. Dividing that profit loss by the emission reduction calculated by the indicator delivers the average net-private ACs per kg CO_2-equ. By stepwise increasing the percentage reductions, points on the AC curve can be simulated. Dividing the change of total ACs between reduction steps by the change in abated GHGs as calculated by the indicator leads to marginal net-private abatement costs (MACs) per unit of CO_2-equ. As stated before, there is a difference between net-private (in-plant) ACs/MACs and net-social (actual) ACs/MACs because the GHG indicator might account for more or less GHGs as actually emitted. The net-social ACs/MACs are calculated by dividing the profit loss induced by the specific applied indicator scheme by the “actual” emission reductions. ”Actual” emission reductions are quantified by using the highest disaggregated indicator scheme as a kind of reference indicator to calculate “real” emission quantities for each optimised farm plan. A detailed explanation of the model approach and calculation procedures of net-private and net-social ACs/MACs is given by Lengers and Britz (2012).

3.2 *GHG indicators*

All implemented GHG indicators 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 a high level of detail and related data needs. The IPCC guideline can be assumed to be scientifically accepted and consistent (to e.g. avoid the multi-accounting bias). We adapted them to German conditions following IPCC (2006) and partly enhanced them by more appropriate literature findings.

In the following, the different indicator schemes are described, briefly explaining how GHG calculations for enteric fermentation, manure management, soil cultivation and fertiliser use are aggregated to whole farm emission-accounting indicators (see Appendix 1 for more detail). This also highlights differences in data demands and how farm attributes and decision variables are impacting the accounted GHG emissions for the indicators. We build on the study of Lengers and Britz (2012) and expand their indicator list by one highly detailed calculation scheme, used as the reference indicator. A more detailed documentation of these indicators can be found in Lengers (2012b).

The simplest indicator is named *actBased* and is equivalent to the Tier 1 methodology from IPCC (2006). Single default emission factors (CO_2, 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). For all indicators, we use background emission factors for agricultural soils from Velthof and Oenema (1997, p.351) for N_2O and from Boeckx and Van Cleemput (2001) for CH_4 which deviate from IPCC (2006)[4].

The second indicator (*prodBased*) is derived from the activity-based one, differing in the calculation of emissions from lactating cows and crop production. The default Tier 1 emission parameters for cows are divided by an average milk yield to derive a default per kg milk emission parameter. Default per ha emission factors from crop production activities are divided by average crop yields to arrive at default per unit of output emission factors. Multiplying these factors by the realised milk or crop yield delivers emissions per cow and ha, which are linearly increasing in yield levels.

The *genProdBased* indicator scheme is slightly more disaggregated compared to the prodBased one. It depicts the effect that emissions per kg of milk actually diminish with increasing milk yield. GHGs from lactating cows rely on emission calculations incorporating gross energy (GE) demand for maintenance, growth, activity and lactation (IPCC 2006). Hence, with increasing milk yield, emissions linked to the cow's energy demand other than for lactation are distributed to a higher output quantity such that the emission factors per kg of milk diminish. 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 fertiliser are assumed.

A more detailed calculation scheme is defined by the *NBased* indicator. Emissions from enteric fermentation are derived by using GE demand according to

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[4] This has to be done because background emission factors from IPCC (2006) are based on a study about peat soils in case of N_2O. CH_4 background emissions from soils are not recognised by IPCC methodology.

IPCC (2006) requirement functions for animals depending on genetic potentials and assuming default feed digestibility. Methane and nitrous oxide emissions are calculated based on monthly manure amount in storages with storage type specific emission factors (subfloor, surface storage with or without coverage...). The emissions from crop production are based on actual monthly N applications, differentiated by synthetic and organic fertiliser, the latter also depending on application technique (broad spread, drag hose, injection).

The most detailed emission indicator is presented by the so called *refInd* scheme. The *reference indicator* represents the indicator with the highest degree of precision in calculating real emissions from the production portfolio of the farms. Thus it is taken as benchmark for the GHG accounting precision of the other indicators. In principle, it is equivalent to the calculation procedure of the NBased indicator, but with some improvements in emission calculation from enteric fermentation. Emissions are quantified based on real GE intake and not on theoretically valid GE demand (to consider the possibility to not fully utilise the genetic potential of the cows). As digestibility of feeds has a particularly high impact on the occurring emissions due to enteric processes (Benchaar and Greathead 2011; Hellebrand and Munack 1995; Machmüller and Kreuzer 1999), the refInd also recognises differences in energy digestibility of the feed supplements. Furthermore, the indicator accounts for the addition of fats and oils to the ration to improve the digestibility. Calculations of emissions from manure management and fertilisation are equivalent to the NBased indicator.

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

3.3 *Abatement options recognised by indicators*

Farmers will only realise abatement options credited by the applied emission indicator, even if other options would be more cost-efficient. As shown in table 1, the design of each indicator as discussed in the paragraph above is linked to a specific selection of abatement strategies. These are taken from Flachowsky and

Brade (2007), Oenema et al. (2001) and Osterburg et al. (2009), and only comprise options not banned by German or European law and supported by scientific findings. Lengers (2012a) provides details regarding these abatement possibilities. As seen below, some of them have a more investment-based character (permanent options) whereas others may be changed flexibly during periods or month (variable options). As permanent options may cause path dependencies, the ability of an indicator to trigger more flexible options may markedly impacts the abatement costs on farm.

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

	actBased	prodBased	genProdBased	Nbased	refInd
permanent					
stable type				x	x
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

By definition, our reference indicator covers all GHG mitigation options implemented in the model, thereby guaranteeing the most inexpensive and flexible abatement on farm level up to the point where the combined abatement potential of all single measures is fully utilised. At the other extreme, the actBased indicator allows GHGs reduction only by reductions in herd size and/or crop hectares, directly inducing losses of the full gross margins of the activities given up.

Generally, with growing detail, indicators reflect more abatement strategies and thus open more possibilities for cost-efficient reactions to GHG ceilings. Less detailed indicators drawing on aggregate farm attributes such as the herd size offer rather limited abatement strategies, in extremes only 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).

3.4 *Derivation of ACs*

In order to derive ACs, allowed total farm emissions are stepwise reduced compared to the GHGs emitted without any emission restriction, the so-called baseline. The total “net on-farm” ACs are equal to the simulated profit loss against the baseline provoked by the implementation of an emission ceiling. That loss is net off measurement, administration, and control costs for the emission quantification. 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 AC 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 MAC curves, simulated here in 2% steps up to a 40% reduction in GHG emissions, differ between indicators for the very same farm.

As stated before, 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 as actually emitted. The net societal ACs/MACs are calculated by dividing the profit loss induced by the specific applied indicator scheme by the “actual” emission reductions. “Actual” emission reductions are quantified by using the most accurate accounting scheme, the reference indicator[5]. That normalization thus renders the indicators comparable by correcting differences in GHG estimates for an identical production plan.

3.5 *Model runs*

To analyse differences between the indicators regarding GHGs accounted, abatement costs as well as cost efficiency of mitigation, differently characterised dairy farms[6] (variation of herd size between 30 and 120 cows with a 10 cow interval and yield level from 4000 to 10000 kg head^{-1} year^{-1} with a stepwise increase by 1000 kg) were simulated by the model DAIRYDYN over a time horizon of 15 years. This resulted in 70 different farms. With regard to the cost efficiency of measures, the longer time horizon is important to render investment based and long term abatement decisions attractive (Del Rio 2008). For the derivation of abatement costs, 20 reduction steps à 2% of baseline emissions were defined which implies a maximal reduction of 40% of baseline GHGs for each farm.

4 Results: Evaluation of the indicators

The above described indicator schemes have to be evaluated regarding the indicator requirements identified at the end of the literature section: accuracy, cost efficiency and feasibility. Whereas we can assume that accuracy increases when moving to more detailed indicators and feasibility decreases, little is known beforehand about influences on abatement costs and the size of these effects which are thus discussed in the following.

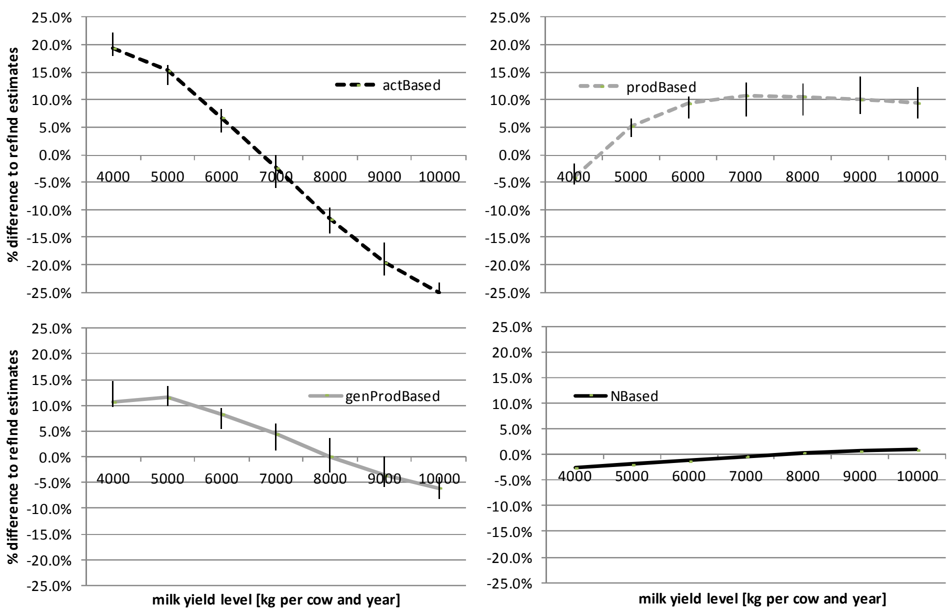
[5] For further detail concerning the normalisation procedure see Lengers and Britz (2012) or Appendix 1.

[6] Farm endowments of land, labour, machinery are adjusted to starting herd size and biological reproduction rates as well as prices are predefined accordant to data collections like KTBL (2010).

4.1 *Accuracy of emission accounting*

Each indicator will typically produce its own emission inventory for the same farm production plan depending on the recognition of process details and implemented emission factors. In order to analyse their accuracy, the emission quantities for each indicator are aggregated over all simulated farms and reduction steps and then related to the results for the reference indicator. The resulting deviations from benchmark estimates of the reference indicator depending on the milk yield of the initial herd are shown in figure 1. The variation in GHG accounting accuracy due to farm size changes is shown by the min and max values (vertical line). The per unit over- or underestimations stay nearly constant, only slightly varying because of changes in the baseline farm plan due to economies of scale in investment decisions for e.g. other manure storage or application techniques.

Figure 1: GHG accounting accuracy of indicator schemes (average % differences compared to benchmark indicator. Variations due to herd size changes are shown by the vertical lines).



Source: own calculation and illustration

Under the assumption that the reference indicator can be taken as the best proxy for real emissions, the NBased indicator has as expected the highest accuracy because it is quite similar. The actBased indicator overestimates emissions for low milk yields up to around 6500 kg per cow and year, with subsequent underestimation growing for output levels up to 10000 kg. The prodBased indicator shows increasing overestimation with increasing milk output levels whereas the genProdBased scheme approaches the benchmark emissions from above with higher milk yield and even underestimates reference emissions by about 6% for a farm with 10000 litre cows.

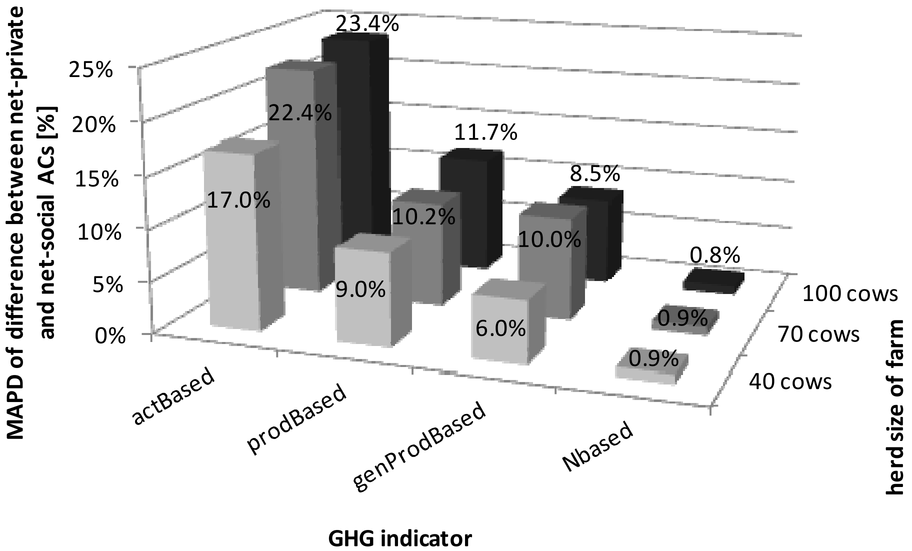
The most detailed emission indicator NBased shows the highest accuracy in GHG calculations with a mean absolute percentage deviation (MAPD) of 1.1% from benchmark emissions over all simulation runs. The genProdBased estimates lead to a MAPD of 6.4%, the prodBased one to 8.5% and the simplest indicator (actBased) to 14.3%.

Accordingly, the higher the aggregation level of indicator relevant variables, the lower the accuracy of emission accounting is because of an increasing sensitivity of measurement bias towards varying farm attributes.

4.2 *Induced abatement costs*

The differently designed indicators lead to different reactions on farm level and induced profit losses when emission ceilings are implemented. This leads to markedly different MACs depending on the indicator applied. Furthermore, the accuracy of calculated GHGs by the indicators influences the differences between net-private and net-social abatement costs, dependent on the amount of under- or overestimation of actual emissions reduced by the chosen abatement options. Figure 2 illustrates the MAPD for the difference (positive and negative) between net-private and net-social average ACs derived over all simulated farm runs and reduction steps (illustrated for three starting herd sizes).

Figure 2: 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

It is obvious that more detailed and thus accurate indicators also show small differences between net-private and actual (net-social) abatement costs. Additionally, simpler indicators might not credit promising abatement options driving up abatement costs.

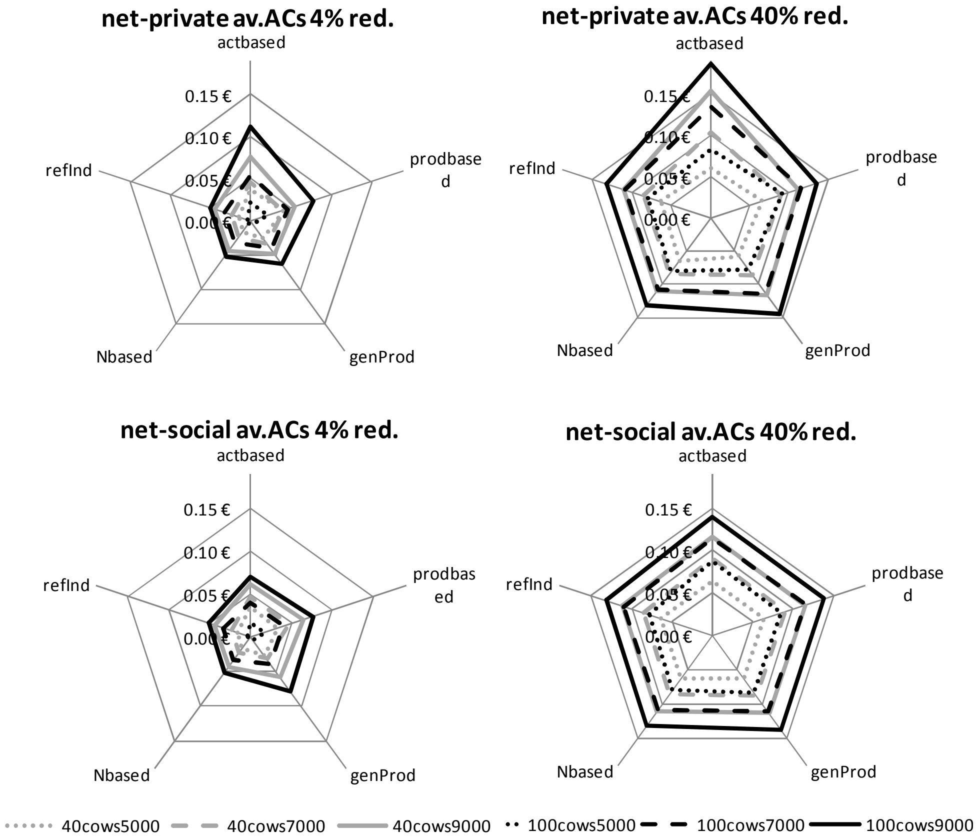
We illustrate selected findings for net private and social abatement costs in figure 3 using simulation results of farms with 40 and 100 cow initial herd sizes and three milk yield levels (5000, 7000 and 9000 kg milk per cow per year). The left-hand figures illustrate the average net-private and net-social abatement costs for a 4% reduction level of baseline GHGs depending on the indicator; the right-hand figures for a total reduction level of 40%. As to be expected, the average ACs at 40% are higher as MACs increase with higher abatement quantities.

Average net-private ACs are highest under the simplest indicator (actBased): up to 112 €/t CO_2-equ. for a 4% and up to 190 €/t CO_2-equ. for a 40% reduction, whereas the most detailed indicator scheme (refInd) leads to the lowest average

ACs. The higher net-private ACs induced by simpler indicators are linked to the fact that cost-efficient abatement options are not credited, see also table 1.

Independent of the indicator chosen, farm attributes have a sizeable impact on mitigation. For example, average net-private ACs vary between 60 and 130 €/t CO_2-equ. depending on farm attributes under the NBased indicators at a 40% GHG reduction.

Figure 3: 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 differences in average net-private ACs between indicators diminish when average ACs are normalised by the reference indicator to net-social costs. The two spider charts in the lower part of figure 3 show that differences in net-social ACs between indicators are considerably smaller than for net-private ones. The normalisation reduces the ACs for simpler indicators, which shows that the example farms actually reduce more GHGs than accounted for by the applied indicator schemes. At the 4% reduction requirement, high differences in average ACs between indicators after normalisation can be observed. These almost completely vanish at 40% GHG reduction, as abatement strategies no longer differ between indicators at higher abatement levels. The cost efficiency of more detailed indicators thus vanishes with increasing abatement.

Figure 3 compares abatement ACs only for very low and quite high reduction requirements. We will now look in more detail into the influence of a stepwise increase in abatement requirement (figure 4), using two rather extreme examples from the spread of analysed farms: a farm with 40 cows and 5000 kg milk yield and one farm with 100 cows and a milk yield potential of 9000 kg in the initial herd.

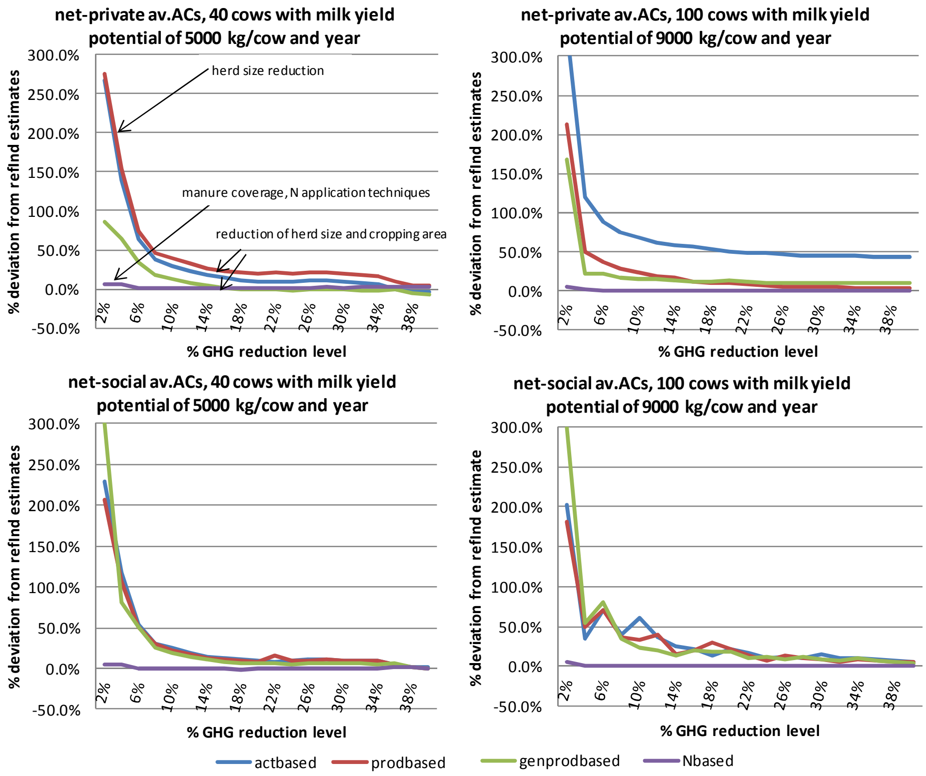
Although indicators account for a various number of different mitigation options, each mitigation option has a maximal level of abatement potential. If the abatement potential of a cost efficient option is fully utilised, the next expensive option will be used. This will lead to increasing MACs and potentially to mitigation strategies for higher abatement levels with little difference between GHG indicators. For example, GHG saving manure handling may cost less per GHG abated than reducing the herd size, but its´ possible contribution to overall GHG abatement is limited. Higher emission ceilings may therefore only be feasible with a reduced herd size, which leads to similar reduction strategies independently of the indicator. Thus, the advantage of more detailed indicators in offering more cost efficient mitigation options will decrease with increasing abatement level.

As already seen from figure 3, at low reduction levels, MACs induced by less-detailed GHG indicators exceed sizeably those of more detailed ones. With increasing GHG reduction, differences in MACs between indicators decrease and hence also the average ACs per kg CO_2-equ. align (dividing the integral below the

MAC curve by the GHG abatement amount). Figure 4 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 4 illustrates differences in net-private average ACs. Obviously, especially for the first reduction steps, net-private ACs induced by the actBased indicator by far exceed those induced by the reference indicator (bearing in mind that net-private abatement costs are related to the GHGs accounted by the specific indicators, not the actual reduction amounts). For the bigger high yield farm shown on the right hand, these differences remain quite high also for consecutive reduction steps.

Figure 4: Net on-farm and net societal average ACs for a 4% and 40% reduction of baseline GHGs depending on indicator scheme, relative to refInd estimates.



Source: own calculation and illustration

However, also for the first farm type shown on the left hand side, average costs for a 20% emission reduction under the actBased indicator are still 11% higher than for the reference indicator. This highlights that simple indicators might considerably drive up GHG reduction costs at farm level compared to highly detailed ones, independent of the reduction level.

The two lower graphics compare the net-social average ACs. All simple GHG accounting schemes provoke very high and similar cost differences compared to the reference indicator at low percentage reduction amounts. For the illustrated 100 cow farm, the average ACs induced by the actBased indicator for a 2% reduction are two times higher than the most cost efficient abatement offered by the reference indicator. For higher reduction levels, the AC cost disadvantage in net-social cost efficiency of simple indicators drops: average net-social ACs align with increasing abatement as similar abatement strategies are chosen.

In monetary terms, the less cost efficient abatement possibilities offered by the actBased indicator lead to higher ACs for the first reduction step of about 85.5 € per year on farm level for the 40 cow farm. Naturally, the bigger the farm, the higher the absolute GHG abatement quantity for a 2% reduction, related costs and cost differences between indicators. The advantage of a detailed indicator for a 2% GHG reduction for the second farm which produces in the starting situation about 4.5 times more milk can be quantified to 495 € per year.

These cost differences between indicators can be explained by the abatement strategies chosen for a 2% reduction. The most cost efficient abatement is achieved by adjusting manure storage times and the implementation of new manure coverage techniques (foil, straw). However, that option is not credited under the actBased indicator where the abatement strategy consists of far more costly herd size reductions and changes in cropping patterns.

For higher abatement levels, the net-private average ACs still differ between indicators and induce strictly higher on-farm costs for simpler indicators. However, when these costs are related to actual abated GHGs to reflect the societal point of view, at higher reduction levels, more detailed indicators lose a bigger part of their advantage in abatement cost efficiency.

5 Discussion

To summarise, the five indicator schemes used for comparison show distinct differences concerning feasibility, measurement accuracy and abatement cost efficiency. The necessary activity or production based data for cropping[7] or animal husbandry[8] needed for simple indicators might be easy to collect and to control, and are often already part of legal reporting obligations of farmers. Information on specific feed ingredients in rations, digestibility of feed supplements and declarations of manure storage times, to give examples of data necessary for complex indicators, are currently not available, will be costly to monitor and might be hard to control, which potentially lowers data reliability (Oenema et al. 2004, p.175). Both farmers and policy makers therefore ask for indicators which draw on reliable and available farm level data which clearly favours simple indicators (e.g. actBased).

On the other hand, the more sensitive an indicator reacts to changes of farm level processes impacting GHG emissions, the more accurate the resulting GHG inventories are (c.f. section around figure 1). Our findings suggest that firstly, the accuracy of indicators is more influenced by milk yield changes than by total farm size variations and, that secondly, the bias of simpler indicators reacts most sensitively to changes in farm attributes (c.f. MAPD of GHG estimates from benchmark). Hence, the use of a simple indicator would lead probably to strong over- and underestimates of actual emission reductions for certain farms, especially

[7] agricultural land registers

[8] Since September 1999, all cattle in Germany are to be reported to the HIT-data pool following § 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.

those with milk yields differing sizeably from the values the simple indicators are calibrated on. Whenever the policy maker is confronted with very heterogeneous dairy farm structures, like in the EU or even in Germany alone (IT.NRW 2012), highly detailed indicator schemes would be preferable to avoid inaccuracies in GHG inventory estimates and an unfair treatment of different farm types, a requirement also raised by Oenema et al. (2004, p:178).

Our findings regarding the abatement costs per kg CO_2-equivalent cannot be directly compared to other studies for a number of reasons; however we will only highlight the most important ones. Firstly, studies in the field publish MACs for different GHG abatement ceilings which might not match the range analysed by us (we showed average ACs). Secondly, we made simultaneous experiments over a wider range of relevant farm attributes without aiming at a consistent weighting reflecting the underlying farm population, whereas others studies use e.g. 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 part of the planned future work. Thirdly, we abate GHGs in sum over a longer decision horizon while allowing for certain investments, which are features often not found in other studies. And finally, the indicators used are not always easily compared to others. Nonetheless, the MAC values simulated by us for different abatement levels are acceptably close to findings e.g. from DeCara et al. (2005), DeCara and Jayet (2006), Durandeau et al. (2010) or Pérez (2005).

Building on the work of Lengers and Britz (2012) we also looked at the differences between net-private and net-social costs which directly depend on the accuracy of the indicator. Simple indicators can provoke quite large differences between net-private and net-social ACs per kg CO2-equ. (c.f. figure 2), reflecting the bias between accounted and actual emissions.

The simulation outputs offer interesting results concerning the relationship between indicators, farm attributes, abatement level and abatement costs, the latter depending on which abatement strategies are triggered by a specific indicator.

Differences between net-private and net-social average ACs per GHG abated strongly vary between farms, especially with different milk yields, and increase with higher abatement levels (see figure 3). Different indicators will hence lead to quite different net-private ACs, and these differences do not vanish with higher reduction levels. A different picture emerges for net-social average ACs: here, deviations on ACs from the reference indicator are quite low at higher reduction levels independent of the indicator applied. This is due to the fact that optimal abatement strategies become quite similar for high abatement levels for all indicators. Accurate abatement indicators that credit cost effective abatement options are advantageous only for small reduction amounts where these options matter. Their advantage levels off with increasing mitigation once the potentials of low cost measures are fully utilised. Once all indicators trigger comparable, more expensive abatement strategies, average net-social ACs are shown to equalise for each single farm, independent of the indicator applied.

In this paper we illustrated our results with only a few chosen example farm types. From these examples no strong conclusions can be taken on the influence of indicator choice for EU or German dairy farms in total to discuss the capability of the GHG indicators in a broader political context. For any further step in that direction, an up-scaling of single farm characteristics to the farm population of the territory in question would be needed.

In order to keep the analysis simple, we have chosen an age of buildings and equipment which ensures that the farms analysed do not need to re-invest in buildings over the planning horizon. Equally, we assumed that neither buildings nor machinery can be sold during the simulation horizon. Otherwise, introducing GHG ceilings might trigger complete farm exit decisions. With a farm exit, GHG emissions would drop completely to zero at a certain GHG emission target and the calculation of marginal or average ACs beyond that point would be useless. The smooth reaction of the model hence reflects the fact that farmers possess capital endowments which they cannot sell. Additionally, we assumed that farm labour is able to work hourly off farm, however at quite low wages.

6 Summary and conclusion

This paper analysed trade-offs between measurement accuracy, abatement cost efficiency and feasibility of five differently detailed GHG emission indicators for dairy farms based on simulations with a highly-detailed single farm bio-economic optimisation model.

Drawing on sensitivity experiments over a wider range of relevant farm attributes, we found that only highly detailed GHG indicators seem to allow for an accurate accounting of emissions both over a large range of emission ceilings and for farms differing in key attributes, such as milk yields, while at the same time allowing farmers to be credited for cost-effective abatement options. In all of the cases analysed by us, more detailed indicators led to lower abatement costs. These detailed indicators require however data on farm process such as the actual feed mix used which provoke quite some reporting burden to farmers and where controlling farmers' notifications might be hard.

Simple indicators can draw on existing farm level reporting data where notifications can be easier controlled, but can lead to highly inaccurate emission estimates. The latter occurs when the default emission parameters underlying the indicator do not fit to process attributes of a specific farm. Implementing simple indicators would probably mean that some farms need to abate more GHGs than they actually would have to, while abatement efforts for others would be overestimated; accuracy is thus also a question of fairness. Equally, simple indicators typically do not credit all cost-efficient abatement options and thus can trigger far higher abatement costs compared to more detailed ones. A lack of accuracy also leads to a difference between private abatement costs – related to GHG emissions accounted by the indicator used – and social abatement costs – related to the actual GHG emissions abated – lowering the precision of any indicator based policy instrument.

A highly interesting and important finding is that the advantage of highly disaggregated indicators to credit more cost-efficient abatement options diminishes with increasing GHG abatement levels.

Thus, when higher emission abatement levels are targeted, less detailed and easier to monitor and control indicators might not provoke sharply increased abatement costs compared to more detailed indicators. Our findings suggests that the accuracy of quite simple indicators which e.g. only relate to herd size and crop areas, can be improved if some relatively easy to monitor or control farm attributes such as the milk yield or manure storage and distribution technology are taken into account. The advantageousness of simple indicators at higher emission levels would increase if monitoring, control and further administrative costs would be included (Osterburg 2004).

Our results indicate that judging an indicator with regard to accuracy, abatement costs and feasibility requires the context of the relevant targeted range of emission reduction. The major advantage of complex indicators is the crediting of cost-effective abatement options which are however hard to monitor and control. An interesting question is thus if other policy interventions could be used to implement these options while using simpler indicators with corrected default emission factor reflecting that these options are already implemented. To give an example: if certain feed additives would be a promising and cost-effective strategy, one could render them compulsory in concentrates for ruminants, instead of using an indicator requiring for farmers to report the exact feed mix they use. This idea can be underlined by the shown effect, that, for higher reduction aims, higher feasibility and applicability can be bought by only small curtailment in abatement cost efficiency.

The above discussed results do not clearly hint at a best indicator to choose. Not only should the targeted emission reduction be relevant, but also monitoring and control costs should be taken into account (Osterburg 2004) for a discussion about the practical applicability of indicators in policy pattern.

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Appendix 1. Indicator schemes