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Soil erosion potential of organic versus conventional farming evaluated by USLE modelling of cropping statistics for agricultural districts in Bavaria

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Abstract. Organic agriculture (OA) aims to identify a production regime that causes less environmental problems than conventional agriculture (CA). We examined whether the two systems differ in their susceptibility to soil erosion by water. To account for the large heterogeneity within the rotations practised on different farms, we chose a statistical evaluation which modelled erosion using the USLE method from the cropping statistics for 2056 districts in Bavaria (70 547 km^2; 29.8% arable). Physical conditions of erosion were determined in a rectangular grid yielding 13 125 grid-cells of c. 5 km^2 each. For validation, erosion was measured in 10 sub-watersheds on two neighbouring OA and CA farms over 8 years (287 erosive events). On average, about 15% less erosion on arable land was predicted for OA than for CA due to the larger area of leys, although OA occupies areas that are susceptible to erosion more often than CA. The same conclusions could be drawn from the validation data. These data also demonstrated that erosion could be reduced considerably below 1 t ha^{-1} yr^{-1} with best management practices under both farming systems. In contrast, at the countrywide scale, cropping did not change adequately with site conditions favouring erosion. The need for erosion control seems not to influence crop rotation decisions on erosion-prone sites.

Keywords: Erosion, crop rotation, USLE, soil conservation, model validation, ley

INTRODUCTION

Soil erosion is regarded as being one of the most serious environmental problems associated with land use (Morgan 1996). In many cases, erosion causes an almost irreversible decline in soil productivity and other soil functions (Biot & Lu 1995; Bruce *et al.* 1995) and leads to environmental damage. For example, the quality of surface water bodies may be adversely affected by translocation of arable topsoil enriched in nutrients and pesticides into adjoining terrestrial and aquatic ecosystems (Verstraeten *et al.* 2002).

Several erosion processes are known, the most important being erosion by flowing water (water erosion), wind ('wind erosion') and soil translocation by tillage ('tillage erosion'). All three damage the soil resource but only the first two additionally cause severe environmental problems because translocated soil leaves the arable area and enters neighbouring ecosystems. Although water and wind erosion are different processes, they are governed by similar principles as far as land use is concerned. Soil surfaces destabilized by tillage and covered with little living or dead biomass are susceptible to erosive forces exerted by air or water. Wind erosion is mainly a problem of coastal landscapes or large plains, while water erosion is of significance more widely. Furthermore, the amount of soil lost by water erosion far exceeds the amount lost by wind erosion in most cases (Heimlich & Bills 1986). Hence, in the following analysis we will concentrate on water erosion, although to some extent our analysis may also hold true for wind erosion due to both processes having similar agricultural impact.

Soil erosion is highly variable in time and space, which makes it difficult to base an assessment on short-term measurements only, for example over several years or on small plots. To overcome this problem many soil erosion models have been developed and are accepted tools for studying soil erosion (Nearing *et al.* 1990). The Universal Soil Loss Equation (Renard *et al.* 1994; Wischmeier & Smith 1978) is one of the oldest models, which is still frequently used. It has a large experimental background, has been adapted to many areas in the world and is still among the best tools for long-term assessment of soil erosion by water (Nearing 1998). The model has been extensively customized over 20 years using data of about 1000 rainfall simulations and 500 plot years under natural rain (summarized in Schwertmann *et al.* 1987) and yielded $r = 0.79$ with measured soil losses on a field scale for six fields covering a total of 232 field years (Schwertmann & Schmidt 1980).

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Organic agriculture (OA) aims to be a production system that is in closer alignment with natural cycles and processes than conventional agriculture (CA). Hence OA should also be less conducive to erosion than CA, although this is yet to be proved. To our knowledge, there is only one study which compares soil loss on a conventional to that on an organic farm and demonstrates a smaller soil loss for OA (Reganold *et al.* 1987). There are also some studies comparing soil properties which influence erosion like infiltrability, aggregate stability or earthworm abundance (Mulla *et al.* 1992; Scullion *et al.* 2002; Mäder *et al.* 2002; Pulleman *et al.* 2003).

However, these attempts may be insufficient to draw firm conclusions for two reasons. First, a quantitative comparison requires that the two types of farm differ only with respect to the farming system while all other parameters match. This condition cannot be met and proven for large landscape elements like fields or farms. This is especially true for soil erosion, which depends on many site properties like soil erodibility, topography and rain erosivity – all known to change over short distances. Second, a quantitative comparison requires that the farming systems under consideration can be clearly defined. While this may be true to some degree for conventional farms in a particular landscape, this premise fails for organic farms. Each farm has to be considered unique owing to diversified strategies to incorporate N-fixing legumes in crop rotations, complicated and diversified rotations, specific adaptations to cope with unfavourable site conditions and exploitation of small market niches. Any extrapolation of inference from an individual organic farm to make a universal generalization about organic farms is thus inappropriate. 'Organic farming' can only be evaluated by taking into account all organic farms. Hence, in this study we compare the degree of erosion on all organic farms in Bavaria to that of all conventional farms in the same region. This comparison is based on modelling, which allows us to specify site influences on erosion when comparing the influence of farming systems.

MATERIALS AND METHODS

Modelling approach

The Universal Soil Loss Equation (USLE; Wischmeier & Smith 1978) predicts long-term average, annual soil loss from the multiplication of six complex terms:

$$A = R \times K \times L \times S \times C \times P \quad (1)$$

where

A long-term average annual soil loss ($\mathrm{t\,ha^{-1}\,yr^{-1}}$),

R rainfall and runoff erosivity ($\mathrm{N\,h^{-1}\,yr^{-1}}$),

K soil erodibility ($\mathrm{t\,h\,ha^{-1}\,N^{-1}}$),

L, S dimensionless topography factors quantifying the influences of the watershed area and watershed curvature,

C dimensionless factor quantifying the influence of the cropping system,

P dimensionless factor quantifying the influence of permanent erosion control measures like terracing and contouring.

The C factor quantifies the influence of cropping. Hence whether the farming system is organic or conventional will especially have an influence on C. The C factor is computed from the combination of the so-called soil loss ratio (SLR) with the erosivity index (EI) (Wischmeier & Smith 1978). The EI quantifies the seasonal distribution of rainfall erosivity. The SLR quantifies the susceptibility of the soil surface relative to the conditions that occur in a freshly prepared seedbed, which is thus considered a standard. The SLR mainly depends on tillage and soil cover. It can be determined experimentally, for example, by rainfall simulator experiments (e.g. Chow & Rees 1994) or by calculation from sub-models (Alberts *et al.* 1989).

The long-term average C factor can only be computed for complete rotations for two reasons. First, between two main crops there is a period, sometimes of several months duration, in which considerable erosion may occur but cannot be assigned either to the previous or to the following crop. Second, carry-over effects exist by which the preceding crop influences the extent of erosion during following years. This is especially true in ley-based rotations. Sod-forming crops like clover-grass are known to stabilize the soil. This decreases soil loss up to two years after the sod has been ploughed as compared to an otherwise identical system without sod (Wischmeier & Smith 1978). These carry-over effects of leys are also identified in other models like the 'prior land use factor' in EPIC (Sharpley & Williams 1990), which is a modification of the USLE, but also in models that use a completely different prediction technology like EUROSEM (Morgan *et al.* 1998) or WEPP (Lane & Nearing 1989). In the latter models the higher organic matter content, the higher aggregate stability, more earthworm channels and lower erodibility (Siegrist *et al.* 1998, Scullion *et al.* 2002, Pulleman *et al.* 2003) after inversion of leys would cause a similar effect.

Data on crop rotations may be raised while examining individual farms. They are not available, however, for larger areas because to our knowledge no statistical inventory of rotations exists. This is true for CA and OA. Therefore, C factors assumed in this study have to be computed on the basis of cropping statistics.

Auerswald (2002) computed C factors of many conventional and organic rotations and combined the rotations in a Monte-Carlo simulation to simulate the effect of a combination of different farms. He showed that the C factor averaged over different farms can be estimated from cropping statistics. The mean absolute error between this estimation and the average from the accurately determined C factors of the rotations was 0.016 only. The C factor may hence be estimated using rather simple parameters with one equation being valid for both farming systems:

$$C = \{[830 - 15.8\,(G+M+S) + 0.082\,(G+M+S)^2]\,(1 - 0.03\,S) + 0.1\,S - 0.5\,M + 27\}/\,1000 \quad (2)$$

where

G is the percentage of small grain (including oil seeds),

M is the percentage of row crops planted in mulch tillage,

S is the percentage of sod-forming crops.

Mulch tillage is the planting of row crops into a mulch cover created by the cultivation of cover crops, which are either frozen down during winter or chemically killed prior to row crop sowing or planting (Kainz 1989). In cases where equation 2 predicts C factors of less than 0.01 the C factor has to be set to 0.01, and where it exceeds 0.45 it has to be set to 0.45 (Auerswald 2002).

Study area

Bavaria is a large state in the southern part of Germany with an area of 70 547 km^2, comprising 29.8% arable land, 16.7% grassland and 34.6% forest It is characterized by a comparatively wide range of site conditions, which allows extrapolation of findings in a study such as this to other German states or to neighbouring countries like Austria and Switzerland (Auerswald 2002). However, compared to larger areas like the United States of America, the range of site properties in Bavaria has to be regarded as limited. As far as soil erosion is concerned, however, conditions encountered in Bavaria lie right in the middle of the range found in the USA (Auerswald 1991).

Bavaria is divided into 2050 districts. For each of these districts average cropping records exist from the INVEKOS inventory (INtegriertes VErwaltungs- und KOntrollSystem zur Kontrolle von flächengestützten Förderanträgen; integrated administration and control system for European Community aid schemes). The INVEKOS inventory covers about 97% of the agricultural area and thus provides accurate information about cropping practice, enabling the average impact to be calculated by summation of the separate impacts of the various cropping rotations. The INVEKOS data were separated into organic and conventional areas and the average C factor was computed for each district based on equation 2 and the respective inventory data.

We used only data from rotations but not from permanent crops like hops, grapes or asparagus and we did not compare the percentage of grassland, which may also deviate considerably between CA and OA. Not accounting for the percentage of grassland should not have affected conclusions of our study because erosion on productive grassland can be regarded close to zero in either case.

Data evaluation

The C factor alone may be insufficient to compare the erosion risk of two farming systems, when accumulations of organic farms in certain regions occur, which may be characterized by non-average erosion conditions. The C factors hence have to be combined with the other factors of the USLE to yield total soil loss. To guarantee a sufficiently high resolution, Bavaria was divided into a rectangular grid of 13 125 cells, each about 5 km^2 in size. For each of the grid cells the R factor was computed according to Rogler & Schwertmann (1981); the K factor was estimated following Auerswald (1986), and the L factor following Mutchler & Greer (1980). The equation by Nearing (1997) was used to compute the S factor from slope gradient. It is suitable also on steep land as found in alpine areas or grape-growing areas. The P factor is of minor importance at the district level. On average it is 0.85 in many regions of Bavaria (Kagerer & Auerswald 1997). Methods of data acquisition and interpretation were described by Auerswald & Schmidt (1986). By multiplying the product $RKLSP$ by the respective C factors of OA or CA of each grid-cell we are able to predict regional and general differences in soil loss between OA and CA systems.

The average cropping conditions on either organic or conventional arable land were computed by weighting each grid-cell according to the proportion of respective arable land:

$$V_{\mathrm{av}} \sum_{i=1}^{13\,125} V_i F_i / \sum_{i=1}^{13\,125} F_i \tag{3}$$

where

V_{av} the average of any variable like rain depth or rain erosivity for arable land of OA or CA,

V_i the value assigned to this variable in each of the 13 125 grid-cells;

F_i the area of arable land found in each grid-cell for either OA or CA.

A statistical evaluation of the difference in the averages between OA and CA is then not possible and not necessary because the averages are not computed from a sample subset of the total population but from the total population itself. Uncertainties or errors in the data cannot be quantified but should be small for such a large data set as long as there is no general bias.

Validation

Validation *sensu strictu* is not possible because long-term field scale measurements of erosion on many farms distributed over the country would be necessary. However, we will use data from two neighbouring farms, one conventional (68 ha) the other organic (43 ha), where soil loss had been continuously measured on a field to sub-watershed scale for 8 years in 10 small sub-watersheds ranging in size from 0.5 to 16 ha. The sub-watersheds consisted of less than one field to a few fields because no artificial borders are allowed at this scale. The 10 sub-watersheds were selected out of 16 to have identical soil use except for the type of farming with 83.7/83.1% arable land, 10.3/10.8% grassland and permanent set-aside, 4.6/5.0% field borders, and 1.4/1.1% farm roads in the conventionally and the organically farmed watersheds, respectively. Runoff and soil loss were measured on an event base by sampling 0.5% of the runoff with Coshocton-type runoff samplers where runoff was concentrated by topography and/or field borders. For details of the measurement and validation of the measuring system, see Fiener & Auerswald (2003a). Soil loss was modelled with high resolution using the USLE (average polygon size: 13 m^2; Fiener & Auerswald 2003a). Best management practices were applied as appropriate to individual farming systems (i.e. optimized field layout with field borders acting as runoff barriers, use of ultra-wide tyres, reduction of field passes, use of intercropping, catch crops and residue management to increase surface cover, use of grassed waterways). For details of management, see Auerswald *et al.* (2000) and Fiener & Auerswald (2003a, b).

'Measured' C factors were calculated from the measured soil loss and the predicted bare fallow soil loss $RKLSP$ of the instrumented watersheds after adjusting the measured soil loss for the effects of the grassed waterways and the retention ponds at field borders, as quantified by Fiener &

Auerswald (2003a). 'Predicted' C factors for the specific rotations of both farms were calculated from biweekly soil cover (plants, residues, stones) measured in 15 fields at three geodetically defined locations over four years. (For examples of data see Auerswald *et al.* 2000.) The SLRs were calculated from soil cover using the equation determined by Kainz (1989) with rainfall simulator experiments under similar conditions. The C factors were then computed from daily SLRs and the annual distribution of the erosivity index (EI) taken from Rogler & Schwertmann (1981). The carry-over effect after inversion of ley on the organic farm was taken from Wischmeier & Smith (1978).

RESULTS AND DISCUSSION

There is a tendency for OA to occupy less favourable arable sites than CA. On average, the arable OA sites receive more precipitation and have soils which are less deep and slopes with steeper gradients (Table 1). The higher annual precipitation corresponds to a greater rain erosivity, R, and the steeper slopes to a greater S factor. The shallower soils of OA are also stonier, sandier or clayier and hence have a 14% lower soil erodibility, K. This difference in K, however, is too small to compensate for the 27% greater R and 15% greater S.

Land use accounts for these more unfavourable conditions by smaller fields leading to 19% shorter erosive slope lengths. The effect on the L factor, however, is only 4% due to the low sensitivity of the L factor to length changes in this range of slope length. Consequently, it does not compensate for the effect of the other site-specific properties. Hence the bare fallow soil loss, $RKLSP$, is about 14% greater on organic arable land than on conventional land (Table 1).

Among all 2050 districts, 85 had no arable area and will not be considered further. OA contributed between 0 and 100% to the arable area. On average it covered 3.6% of the arable land (Table 2). Hence, it has a negligible influence on the district-wide average soil loss. The distribution between districts is very uneven, which is demonstrated by the 25% quartile being equal to 0% and the 75% quartile being equal to the mean because the mean is largely influenced by a few districts with exceptionally high percentages of OA.

The percentage of small grain on arable land is similar for CA and OA. Distinct differences occur in the percentages of row crop and sod-forming crops between the systems. Predominantly, OA is characterized by a large percentage of grass/legume ley, which is about three times greater than that for CA. This is mainly at the expense of row crops. This difference results from the need to use symbiotic N-fixation of legumes as a source of N instead of mineral N fertilizers, and to control weeds with repeated mowing.

From the wide range in crop areas, a wide range in C factors was expected and observed (Figure 1). The C factors in both systems ranged from 0.01 to 0.45. The low values are restricted to regions of more than 1000 mm annual precipitation growing almost entirely grass. C factors of 0.45 indicate maize monocultures, which are restricted to the same regions because the maize serves to supplement the fodder from grassland. In both cases, the respective districts can be regarded as unimportant outliers in respect to the amount of arable land, although they strongly influence the visual assessment of Figure 1. The 25% and 75% quartiles (Table 2) provide a more realistic picture of the range.

The comparison of the mean C factors (Table 2) predicts that OA reduces soil loss by about 24% as compared to CA under identical site conditions. Due to the more erosive site conditions of OA, the soil loss is only 15% less for OA than for CA. The regression between the C factors of OA and CA, however, only yields $r^2 = 0.128$ (for $n = 1072$ districts), indicating that the rotations and thus the C factors of both systems are governed by different influences. Given the large variability of both systems and the low correlation we should not conclude that OA decreases soil erosion,

Table 1. Average site conditions on arable land of conventional and organic farms computed from 13 125 grid-cells weighted according to their percentage of conventional or organic arable land.

Variable	Unit	Average, conventional farms	Average, organic farms	Difference between organic and conventional farms
Precipitation	mm yr^{-1}	767	981	+28%
R factor	N h^{-1} yr^{-1}	65.4	83.2	+27%
Soil depth	m	0.68	0.64	–6%
K factor	t h ha^{-1} N^{-1}	0.37	0.32	–14%
Slope length	m	185	150	–19%
L factor	None	2.41	2.30	–4%
Slope gradient	%	7.5	8.7	+16%
S factor	None	0.87	1.02	+17%
$RKLSP$[a]	t ha^{-1} yr^{-1}	46.9	53.4	+14%

[a]Quantifies the site-specific soil erosion potential exclusive of the influence of cropping.

Table 2. Comparison of organic and conventional agriculture based on district records for 2001.[a]

		Organic farms (%)	Conventional farms (%)
	25% quartile	0.0	96.4
Total arable land	Average	3.6	96.4
	75% quartile	3.6	100.0
No. of districts		1965	1965
	25% quartile	45.4	48.4
Small grain	Average	56.9	57.6
	75% quartile	73.7	70.7
No. of districts		1081	1956
	25% quartile	7.0	23.1
Row crops	Average	20.8	34.5
	75% quartile	27.4	43.0
No. of districts		1081	1956
	25% quartile	1.6	1.6
Grass/legume ley	Average	22.3	7.9
	75% quartile	32.2	10.1
No. of districts		1081	1956
	25% quartile	3.3	10.1
C factor	Average	9.9	13.3
	75% quartile	11.9	15.9
No. of districts		1081	1956

[a]There were 2050 districts in Bavaria of which 1965 had arable land >0.00% of total agricultural land; 1081 had organic arable land >0.00%; and 1956 had conventional arable land >0.00%.

although this was true for the majority of districts. For 255 out of 1072 districts the C factor of OA was larger than that of CA. The large scatters show that for both systems the C factor, and thus soil loss, can be decreased. In both cases, the expansion of grass/legume leys would decrease soil loss far more than changing from row crop to small grain. Opportunities to increase the percentage of grass/legume leys should hence be explored in both systems.

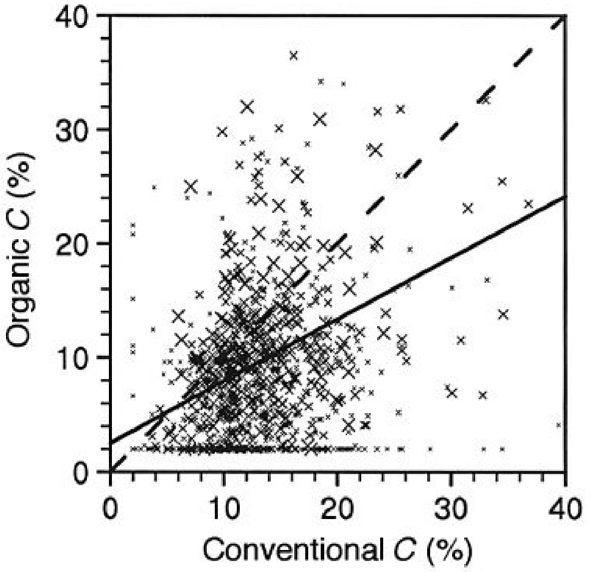

Figure 1. Comparison of C factors (in % of bare fallow soil loss) between organically and conventionally farmed arable land. District averages of 1072 districts in Bavaria with >0.00% of arable land for both organic and conventional farms; marker size increases with increasing geometric mean of arable land for organic and conventional farms; dashed line indicates unity; solid line is the regression $Y = 2.5 + 0.54X$; $r^2 = 0.128$. (Regressing X on Y gives $X = 10.9 + 0.24\,Y$.)

For both systems, the average C factor decreases with increasing site specific erosion potential, which is quantified by *RKLS*:

$$C_{OA} = 13.7 - 1.1 \ln (RKLS) \quad r^2 = 0.020 \quad (n = 6879) \tag{4}$$

$$C_{CA} = 17.1 - 1.0 \ln (RKLS) \quad r^2 = 0.043 \quad (n = 10\,070) \tag{5}$$

These relationships are very weak, however. The C factor decreases only as logarithm of *RKLS* whereas a linear decrease would be necessary to compensate for the effect of *RKLS* on soil loss. Slight changes in cropping thus cannot compensate for the large differences in site-specific erosion potential. This is true for both farming systems. Neither OA nor CA adequately takes into account the site-specific erosion potential in cropping decisions.

The equation used to estimate the C factor (equation 2) accounts for the effects created by the rotation. Organic farming differs not only in rotation but also in the use of agrochemicals. In addition to the rotation effect, absence of pesticides is sometimes claimed, which may additionally influence the C factor. These effects on erosion, however, have not been quantified despite attempts to do so (e.g. Auerswald 1995) and thus have not been considered. However, it is unlikely that their effect is large, otherwise it would have been easy to quantify.

Validation

To our knowledge, the validation data set we have used is the largest available comparing field-scale soil loss from OA and CA. During the 8-year measuring period 287 events induced runoff and soil loss in at least one of the 10 sub-watersheds. The data set, however, represents a unique situation and thus the absolute values of the different parameters for both farms differ from the country averages.

Table 3. Average cropping conditions and erosion parameters on arable land (AL) of the conventional and the organic farm used for validation.[a]

Variable	Unit	Conventional farm	Organic farm	Difference between organic and conventional farms	
				Validation (%)	All (%)
Small grain	% of AL	50.0	42.9	–7	–1%
Row crops	% of AL	50.0	20.8	–29	–14%
Ley	% of AL	0.0	22.3	+22	+14%
Precipitation	$mm\,yr^{-1}$	804	804	0	+28%
R factor	$N\,h^{-1}\,yr^{-1}$	69	69	0	+27%
K factor	$t\,h\,ha^{-1}\,N^{-1}$	0.42	0.32	–24	–14%
No. of fields	n.a.	7	14	+100	n.d.
Field size	ha	4.3	2.2	–49	n.d.
Slope length	m	159	112	30	–19%
L factor	n.a.	2.69	2.25	–16	–4%
Slope gradient	%	8.9	10.4	+16	+16%
S factor	n.a.	1.04	1.36	+32	+17%
RKLSP [b]	$t\,ha^{-1}\,yr^{-1}$	68.9	57.4	–17	+14%
Measured soil loss[c]	$t\,ha^{-1}\,yr^{-1}$	0.29	0.04	–86	n.d.

[a]High-resolution erosion modelling with a total of 52 894 cells; [b]quantifies the site-specific soil erosion potential exclusive of the influence of cropping; [c]measured soil loss is based on 2296 watershed events.
n.a. = not applicable; n.d. = not determined.

Nevertheless, the relative differences between OA and CA in this data set confirm the results from the country statistics (Table 3). The organic farm had more leys at the expense of row crops, which was similar to the countrywide averages. The site conditions exhibited similar differences to the country average except for long-term average precipitation and the *R* factor which are considered identical for two farms separated only by a farm road. The organic farm had soils with a smaller *K* factor. It was situated on steeper land and the *S* factor was larger, whereas the field size and the *L* factor were smaller. The bare fallow soil loss, *RKLSP*, of the organic farm in the validation study was, however, about 17% smaller due to the identical *R* factor. Finally, the measured soil loss from the organic farm was much less than the difference in *RKLSP* (86% vs 17%), proving the lower erosion risk of land use that incorporates leys.

On the conventional farm the *C* factor determined from measured soil loss was 0.036, which matches the *C* factor predicted from soil cover measurement (0.040; range 0.028–0.049). On the organic farm the measured soil loss gave a much lower *C* factor (0.004), while from soil cover alone a higher *C* factor than on the conventional farm was obtained (0.049) because more frequent tillage reduced soil cover. Including the carry-over effect into the prediction yielded a lower *C* factor than on the conventional farm (0.032), but it was still considerably higher than that derived from the measured soil loss. This could be the result of an additional effect of organic farming reflecting the absence of mineral N fertilizers and synthetic pesticides, which until now has not been quantified. However, rigid quantification of this additional effect may not be possible, even with our data set, due to the 8-year measuring period, which covers only one rotation on the organic farm, the limitation to one farm and the error propagation with a multitude of variables.

The measured soil losses on both farms were much smaller than what could be expected on average for all conventional or organic farms. This is due to the adoption of best management practices on both farms (Auerswald *et al.* 2000; Fiener & Auerswald 2003b), which have considerably lowered the soil loss. For one field, which now belongs to the organic farm, Schimmack *et al.* (2002) quantified the soil loss by using atomic-weapon fallout plutonium. Soil loss by sheet and rill erosion (not including tillage erosion) was more than two orders of magnitude greater than after the best management practices were introduced. Averaged over a 23-year period the rate of soil loss was $14\,\mathrm{t\,ha^{-1}\,yr^{-1}}$ compared with substantially less than $1\,\mathrm{t\,ha^{-1}\,yr^{-1}}$ after the land use change.

CONCLUSIONS

In Bavaria, organic farms tend to occupy the more unfavourable arable sites, which are also more at risk of erosion. The estimated site-specific bare-fallow soil loss is hence 14% greater for OA than for CA.

The district average in the proportion of row crops, small grains and grass/legume leys differed greatly between OA and CA indicating that natural site properties have little influence on rotations.

On average, OA will cause about 24% less erosion then CA under otherwise identical site conditions. This can be attributed to the larger area of grass/legume ley, which has the potential to reduce erosion markedly, even two years after inversion. The lower *C* factors more than compensate for the unfavourable site conditions. Hence, the average soil loss is about 15% less for OA than for CA.

There are large deviations on both sides of the average *C* factor indicating that erosion in both farming systems could be reduced considerably. Erosion control does not seem to influence management decisions on crop rotation in either farming type. The lower erosion in OA on average has hence to be regarded as accidental. It is a consequence of the shortage of N supply and the need for weed control, which are partly met by a greater proportion of grass/legume leys in organic rotations (Watson *et al.* 2002; Berry *et al.* 2002). The large effect of the best management practice on the soil loss in the validation exercise also demonstrates that both farming systems have much scope to reduce soil losses.

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