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How to estimate concentrations of pesticides in surface water semi-quantitatively: an interim report

(in addition to EPPO bulletin 23, 1993)

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PREFACE

This report presents a semi-quantitative tool or module on the estimation of concentrations of pesticides in surface water. It pretends to give guidance in comparing — per pesticide — the *potential* contribution of various routes of dissipation to concentrations in surface water. It does not estimate *actual* concentrations. Also, it does not reflect the current regulatory practice with respect to the admission of pesticides in The Netherlands.

Furthermore, this report reflects some of the discussions within RIVM on the feasibility of such semi-quantitative tools for pesticide (re)registration authorities *versus* the scientific validity. Therefore the authors hope that — as an interim report — it will contribute to further discussions.

The content of this report is closely linked with the decision-making scheme for surface water in EPPO (1993). It could be seen as an *addendum* .

TABLE OF CONTENTS

SUMMARY

This report offers a semi-quantitative tool to estimate concentrations of pesticides in surface water due to various routes. The routes via runoff and erosion after rain storms are particularly focussed upon. By running this module — via a simple spreadsheet — the *potential* contributions of runoff and erosion can be assessed. The module does not estimate *actual* concentrations. The basic concept of this module consists of two aspects: first, hydrophobic pesticides are assumed to reach adjacent surface water by the translocation of soil particulates, onto which the pesticide has been sorbed; second, hydrophylic pesticides are assumed to reach adjacent water by the translocation of water on the soil or subsoil, in which a substantial fraction has been solved. This semi-quantitative approach has the advantage of being clearly structured and very feasible. It is also a method to standardise ranking pesticides on their dissipation *potential*. On the other hand, it includes many simplifications with respect to meteorological, (geo)hydrological, pedological, and agricultural aspects. A major feature of the module is that it is primarily based on US research, therefore including parameters, that may not be apt for e.g. Europe. The core of the module is a modified USLE (Universal Soil Loss Equation). There are almost no field data available for validation. The tracking of these may be a goal for a next project. Some data indicate that the module overestimates actual concentrations under field conditions. On the other hand, it is indicated that the *potential* contributions of runoff and erosion may exceed the *potential* contribution of other routes, e.g. drift, substantially.

This report is *ad interim*. It does not reflect current regulatory practice with respect to the admission of pesticides in The Netherlands. It can be seen as an *addendum* to EPPO bulletin 23 (1993).

SAMENVATTING

Dit rapport beschrijft een semi-quantitatieve module om concentraties in het oppervlaktewater te schatten uitgaande van verschillende routes. Het accent ligt hierbij op de bijdragen van de runoff (op of vlak onder het bodemoppervlak) en de erosie na zware regenbuien. Met behulp van deze module — in een spreadsheet — kunnen de *potentiële* bijdragen van zowel de runoff als de erosie worden ingeschat. De module schat *geen actuele* concentraties. De module is gebaseerd op twee complementaire processen: enerzijds worden hydrofobe pesticiden verondersteld het oppervlaktewater te kunnen bereiken door het transport van bodemdeeltjes, waaraan de pesticiden zijn gesorbeerd; anderzijds kunnen wateroplosbare pesticiden het aangrenzende oppervlaktewater bereiken met de waterstroom zelf, waarin de pesticiden zijn opgelost, hetzij ter hoogte van of vlak onder het maaiveld. Elk pesticide wordt derhalve verondersteld het water te kunnen bereiken, deels via de ene, deels via de andere weg. Deze semi-quantitatieve benadering heeft als voordelen een heldere structuur en een makkelijke toepasbaarheid. Bovendien biedt het een gestandaardiseerde manier om pesticiden te prioriteren op hun *potentiële* verspreidingsroutes. De module bevat echter veel simplificaties van diverse complexe aspecten op het gebied van de meteorologie, de (geo)hydrologie, de pedologie en de agronomie. Een belangrijk aspect van de module is verder dat de vergelijkingen vooral zijn gebaseerd op Amerikaans onderzoek, als gevolg waarvan diverse parameters in de module opgenomen zijn die mogelijk niet geschikt zijn om Europese condities te omschrijven. De kern van de module is een aangepaste USLE (Universal Soil Loss Equation). Er zijn nauwelijks veldgegevens beschikbaar om de module te valideren. Het opsporen hiervan zou een van de doelen van een volgend project kunnen zijn. Sommige veldgegevens lijken aan te geven dat de module actuele concentraties in het veld overschat. Aan de andere kant lijkt het erop dat de *potentiële* bijdragen van runoff en erosie substantieel groter kunnen zijn dan die van andere routes, bijvoorbeeld drift.

Dit rapport is *ad interim*. Het geeft de huidige praktijk wat betreft de toelating van pesticiden in Nederland niet weer. Het kan gezien worden als een aanvulling op EPPO bulletin 23 (1993).

1. INTRODUCTION

1.1 General

The European and Mediterranean Plant Protection Organization (EPPO) and the Council of Europe (CoE) have developed guidelines for environmental risk assessment of pesticides (EPPO, 1993). This has been done in advance of current attempts on a European level to harmonize the various existing assessment schemes (e.g. the Uniform Principles). The proposed guidelines of the EPPO/CoE have been prepared by a Joint Panel on Environmental Risk Assessments in the period 1989 - 1993. They include several decision-making schemes suitable for adoption by individual regulatory authorities. Not all schemes, however, had been completed at the time of publication. This report fills one of these gaps. It provides the formulas and expressions necessary for estimating the concentration of pesticides in surface water. Therefore, this report should be seen as additional to the decision-making scheme related to contamination of surface water, presented in EPPO (1993).

This report claims to provide a semi-quantitative tool — or module — on the estimation of concentrations of pesticides in surface water. In this way, the *potential* contributions of the various routes of loading can be assessed. The semi-quantitative tool will be referred to as *module* in this report. The primary routes of loading are considered to be drift, runoff, erosion, overspray, advection in water, hydrolysis, volatilisation, photodegradation, sorption, and biodegradation. The calculations presented in this report, can be used for prioritisation, before triggering further *in situ* investigations.

The generated data should be interpreted very cautiously: they do not pretend to predict actual pesticide concentrations in surface water. They primarily offer a standardised way for comparing pesticides on their loading *potential*. This semi-quantitative approach has the advantage of being clearly structured and very feasible. On the other hand, this approach includes many simplifications with respect to meteorological, (geo)hydrological, pedological, and agricultural aspects.

Routes of loading can be split up in two parts: the first dealing with the processes resulting in loading of the water (nrs 1 - 4 in Figure 1), the second dealing with the dissipation in water after loading (nr 5 in Figure 1). The processes of loading the surface water, are dealt with in the sections 3.1.1 (drift), 3.1.2 (runoff), 3.1.3 (erosion by water), and 3.1.4 (overspray). The processes of dissipation, after loading, are represented in the following sections: 3.2.1 (advection), 3.2.2 (hydrolysis), 3.2.3 (volatilisation), 3.2.4 (photodegradation), 3.2.5 (sorption), and 3.2.6 (biodegradation). Some processes that might be relevant are not included in this report: discharge via drains, atmospheric deposition, accidents (e.g. illegal cleaning of spray equipment), sedimentation, and resuspension.

The emphasis in this report is on the contribution of runoff and (water) erosion to concentrations in surface water (see e.g. calculations in section 3.3). The procedures are reported in Chapter 2, results (i.e. the listing of the relevant formulas and expressions, and some calculation examples) in Chapter 3, and discussion in Chapter 4.

1.2 Runoff and erosion

Following application on soil or plant foliage, pesticides are exposed to a number of hydrological and physico-chemical processes that induce the compounds ' movement and reduction from the soil surface. These processes are strongly related with the meteorological characteristics of the site, in particular with the amount and intensity of precipitation.

The amount of runoff is difficult to predict, as it is assumed to be dependent on various meteorological, agricultural, pedological, and hydrological aspects: rainfall intensity, rainfall distribution, the possibility of ground water recharge, temperature (e.g. the presence of frozen soil), the temporary storage at the surface of the soil, the ground water level, the infiltration capacity, the slope, the type of crop, the type of drainage, and the texture of the top soil (Thunnissen, 1987; Meinardi et al., 1994). It was e.g. shown that the infiltration capacity is primarily related with the type of land use, rather than with specific soil characteristics (Thunnissen, 1987).

In principle, a pesticide can be transported in three directions: upward due to vaporisation from soil or plant surfaces, downward by percolation into the root zone and connected soil layers, and horizontally by overland transport, i.e. by runoff and (water)erosion. In highly arid and sparsely covered areas wind erosion may also occur. Mostly, compartmental soil models that are used for screening purposes do not include estimation methods for overland transport of chemicals and are restricted to modelling the partition and percolation in the soil system.

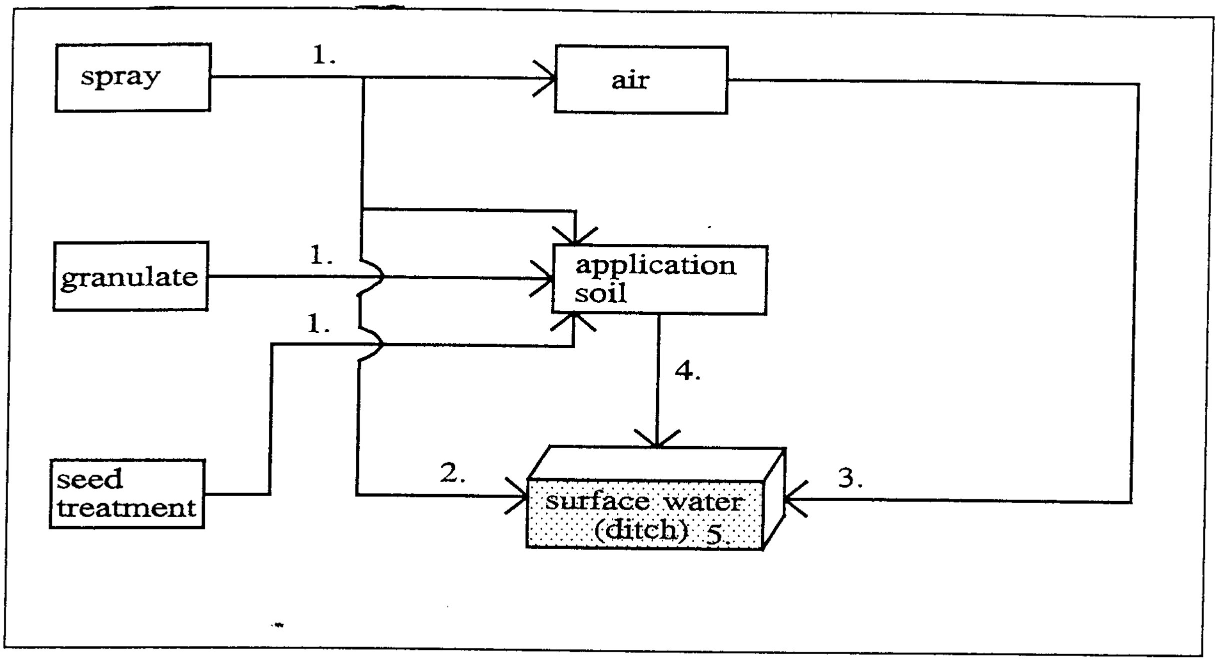
However, it has been recognized that after rain events the contamination level of pesticides in surface waters may increase due to the overland transport. In most cases transfer by runoff and erosion will be the sole inputs of these compounds to surface water systems in an agricultural area, apart from losses during application of pesticides (drift and overspray) and transport in drains. Therefore, it is of importance to incorporate at least a semi-quantitative indication of the extent of the contribution of washed out pesticides to surface water pollution.

Customarily, only runoff is considered as an appropriate transport process in modelling the fate of chemical compounds in the interface of the soil compartment. However, on sloping watersheds also erosion may take place. The energy of the downpour may disengage particulates in the top layer of soil and wash this sediment downhill where it can precipitate into surface water.

In a way these two processes offset each other. Compounds with a relatively high water solubility and a low sorption potential (low $K_{s/l}$) are liable to redistribute themselves with the rain water and will add to the aqueous concentration of the receiving surface water. On the other hand, compounds that demonstrate a high affinity (high $K_{s/l}$) with soil constituents, in particular the organic phase, will likely be transported by the erosion movement and partition to the sediment once entering the surface water. Therefore, in situations where erosion is considered to be of relevance, care has to be taken to associate the transport of the pesticides not only to water solubility (based on K_{ow} or $K_{s/l}$ estimates): wet precipitation may bring about the transport of the compound both in its dissolved and sorbed state. Also, it appears that eroded sediments are enriched with organic matter and that therefore the compound may be redistributed accordingly, resulting in increased concentrations in the erosion flux.

Customarily, mathematical models are used as a standard approach to predict runoff and erosion. However, calculations of overland transport mostly suffer from the limitations and scarcity of input data that are needed to use existing mathematical runoff and erosion models in a practical and reliable way. Moreover, within the scope of the module method presented in this report, it can be understood that the input data should reflect hydrological, meteorological, pedological and agronomic conditions in Europe.

Figure 1. Relevant routes for contamination of surface water



2. PROCEDURES

The equations listed in section 3.1 and 3.2 are based on literature research. They are for the larger part extracted from Carsel et al. (1984) and Ven Te Chow (1964). The erosion equations are based on the Universal Soil Loss Equation (USLE). The equations are listed roughly in a sequence that is adequate for immediate calculation.

The calculations in section 3.3 have been performed with a LOTUS 123© (LOTUS Inc.) spreadsheet.

3. RESULTS

3.1 Loading of surface water

3.1.1 Drift

The equations 1 and 2 can be used to estimate the initial concentrations of a pesticide in water due to drift. A synopsis of the drift fractions F_d is presented in Annex 2. These values are used by the Dutch (re)registration authorities to estimate environmental risks with USES 1.0, an integrated model for risk assessment (EPPO, 1993; RIVM, VROM & WVC, 1994).

$$A_d = 100 \times F_d \times X \times n \qquad (equation\ 1)$$

A_d = *amount entering surface water due to drift* $[mg.m^{-2}]$
F_d = *fraction drift (see Annex 2)*
n = *frequency of application*
X = *dosage* $[kg.ha^{-1}]$

$$C_{dr} = \frac{A_d}{h} \quad (equation\ 2)$$

C_{dr} = *concentration in water due to drift* [$mg.m^{-3}$]
A_d = *the amount entering the surface water due to drift* [$mg.m^{-2}$]
h = *water depth* [m]

3.1.2 Runoff

The equations 3 - 13 can be used to estimate the concentration of a pesticide in surface water, due to runoff. They are listed in a sequential order to facilitate the calculations. All runoff equations refer to a *single* storm event. Therefore indications of time are not included. The rainfall erosion index EI in equation 3 reflects the kinetic energy of a heavy rain storm, causing runoff. Values of I_{30} can be obtained from meteorological stations.

$$EI = 210 + (89 \times \log_{10}(I_{30})) \quad (\textit{equation } 3)$$

EI = *rainfall erosion index* [*cm*]
I_{30} = *maximum 30 minutes rainfall intensity* [*cm*]

The conversion of the rainfall erosion index EI from English to metric units requires the denominator of 100 in equation 4. This conversion renders the runoff factor R_r.

$$R_r = \frac{EI}{100} \quad (\textit{equation } 4)$$

R_r = *runoff factor* [*cm*]
EI = *rainfall erosion index* [*cm*]

The peak storm runoff Q_p in equation 5 is the potential amount of runoff due to a rain storm. This amount is a function of the runoff factor R_r, the area of the watershed A, and type of vegetation cover WCV. The latter implies roughly that a

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more dense vegetation decreases the potential amount of runoff. A synopsis of some WCVs is presented in Table 2 of Annex 1.

$$Q_p = \frac{R_r \times A \times WCV}{100} \qquad (equation\ 5)$$

Q_p = *the peak storm runoff* $[m^3]$
WCV = *Watershed Cover Value* [-]
R_r = *runoff factor* $[cm]$
A = *area of the watershed* $[m^2]$

The potential amount of runoff water minus the potential amount of infiltration renders an estimation of the actual amount of runoff. The potential infiltration S in equation 6 is estimated via the runoff curve number RCN. This RCN is linked with the infiltration rate, some general hydrological conditions — independent of slope and vegetation cover —, and the type of land use. A synopsis with some RCNs is presented in Table 5 of Annex 1.

$$S = \frac{\left(\frac{1000}{RCN} - 10\right)}{100} \qquad (equation\ 6)$$

S = *potential infiltration in a soil* $[m]$
RCN = *Runoff Curve Number* [-]

An estimation of the actual amount of runoff is obtained by equation 7.

$$Q_r = \frac{(Q_p - 0.2S)^2}{Q_p + 0.8S} \qquad (equation\ 7)$$

Q_r = *the estimated actual runoff depth* [m]
Q_p = *the peak storm runoff* [m^3]
S = *potential infiltration in a soil* [m]

After the actual amount of runoff water had been estimated, the actual amount of a pesticide that is assumed to be transported in this runoff water should be estimated. Primarily, one may calculate the volumetric concentration of the pesticide after application (see equation 8). Then it is assumed that there is no interception by the crop, and that the fraction emitted into the air is negligible. Under these worst case conditions all pesticide sprayed is assumed to reach the soil.

$$C_{vol,T} = \frac{X}{10 \times h} \qquad (equation\ 8)$$

$C_{vol,T}$ = *the volumetric total concentration of the pollutant* [$mg.dm^{-3}$]
X = *the dosage* [$kg.ha^{-1}$]
h = *soil depth over which homogeneous mixing of the pollutant is assumed* [m]

After reaching the soil, a part of the pesticide is sorbed onto particular soil constituents as e.g. humic or fulvic acids, or clay minerals. These sorption processes decrease the availability for runoff. Therefore the actual amount of the pesticide in the water should be estimated by a correction for the sorbed amount. The latter can be done via the sorption coefficient $K_{s/l}$, that can be estimated empirically from the octanol water partition coefficient (equation 9) or determined experimentally, e.g. via a batch slurry test. In the experiment, the $K_{s/l}$ equals K_{oc} times f_{oc} or K_{om} times f_{om} ($K_{om} = K_{oc}/1.7$). The empirical equation 9 is according to Karickhoff (1981).

$$K_{s/l} = 0.41 \times K_{ow} \times f_{oc} \qquad \textit{(equation 9)}$$

$K_{s/l}$ = *the soil water partition coefficient* [$dm^3.kg^{-1}$ *soil*]
K_{ow} = *the octanol water partition coefficient* [-]
f_{oc} = *fraction organic carbon in soil (equals the organic matter fraction in soil divided by* 1.7) [-]

Subsequently, the pesticide concentration in the runoff water can be estimated by equation 10.

$$C_{vol,W} = \frac{C_{vol,T}}{1 + K_{s/l}} \quad \textit{(equation 10)}$$

$C_{vol,W}$ = *the volumetric concentration of the pollutant in the aqueous phase of the soil* [$mg.dm^{-3}$]

$C_{vol,T}$ = *the volumetric total concentration of the pollutant in the soil* [$mg.dm^{-3}$]

$K_{s/l}$ = *the soil water partition coefficient of the pollutant* [$dm^3.kg^{-1}$]

The actual runoff flux — dependent on the estimated actual runoff depth, the area, and the concentration in the runoff water — can be calculated by equation 11.

$$J_r = Q_r \times C_{vol,W} \times A \quad \textit{(equation 11)}$$

J_r = *the estimated actual flux in the subsoil due to runoff* [g]

Q_r = *the estimated actual runoff depth* [m]

$C_{vol.W}$ = *the volumetric concentration of the pollutant in the aqueous phase of the soil* [$mg.dm^{-3}$]

A = *area of the watershed* [m^2]

The total volumetric amount of runoff water that is estimated to run down the watershed is calculated with equation 12.

$$V_r = Q_r \times A \qquad (\textit{equation } 12)$$

V_r = *the water volume in the subsoil subject to runoff* $[m^3]$
Q_r = *the estimated actual runoff depth* $[m]$
A = *area of the watershed* $[m^2]$

In conclusion the concentration of the pesticide in the adjacent ditch due to runoff is estimated by equation 13.

$$C_r = \frac{J_r}{V_r + V_d} \qquad (\textit{equation } 13)$$

C_r = *the estimated concentration in the receiving water due to runoff* $[mg.l^{-1}]$
J_r = *the estimated actual flux in the subsoil due to runoff* $[g]$
V_r = *the water volume in the subsoil subject to runoff* $[m^3]$
V_d = *the water volume in the receiving water* $[m^3]$

3.1.3 Erosion

All following equations on erosion refer to erosion due to a *single* rain storm. The time for the water — after a rain storm — to flow from the top of the watershed to the adjacent ditch at the lower side, is estimated by equation 14, an empirical fit. This concentration time is dependent on the length and the height of the slope of the watershed.

$$T_c = \frac{(3.28L)^{1.15}}{7700(3.28H)^{0.38}} \quad \textit{(equation 14)}$$

T_c = *the time of concentration (i.e. the time the water takes to flow from the most distant point of the watershed to the outlet)* [*day*]
L = *the slope length of the watershed (i.e. perpendicular on the outlet)* [*m*]
H = *the elevation of the watershed* [*m*]

The rainfall rate Q_s, e.g. the one-year frequency storm event rate, now can be related with the rainfall intensity I_{30} and the duration of the erosion event T_c (see equation 15). Values of I_{30} can be obtained from meteorological stations.

$$Q_s = \frac{I_{30}}{T_c} \quad \textit{(equation 15)}$$

Q_s = *the estimated peak storm rainfall rate* [$cm.day^{-1}$]
I_{30} = *the maximum* 30 *minutes rainfall intensity* [*cm*]
T_c = *the time of concentration (see equation* 14) [*day*]

The slope-steepness or topographic factor L_s in equation 16 reflects the direct effect of the slope length of the area, and its steepness on the soil erosion by rain. Actually, it is the expected ratio of soil loss per unit area from a field slope with a length — from top to the ditch — of 21.8 meter, and a slope of 9%. The L_s for a bare fallow with such a slope and length is 1.0. The equation had been derived by Wischmeier and Smith (1978).

$$L_s = \left(\frac{L}{22.1}\right)^m \times \left(65.41\left(\frac{H}{L}\right)^2 + 4.56\left(\frac{H}{L}\right) + 0.065\right) \qquad (\textit{equation } 16)$$

L_s = *the slope-steepness or topographic factor* [-]
L = *the slope length of the watershed*
(*i.e. perpendicular on the outlet*) [*m*]
H = *the elevation* [*m*]
m = *elevation factor*; $e = 100 \times H \times L^{-1}$;
$m = 0.2$ *if* $e \leq 1$,
$m = 0.3$ *if* $1 < e < 3$,
$m = 0.4$ *if* $3.5 \leq e \leq 4$,
$m = 0.5$ *if* $e \geq 5$ [-]

Equation 17 is the erosion prediction equation proposed by Wischmeier and Smith (cited in Troeh et al., 1980). This USLE (Universal Soil Loss Equation) pretends to predict long-term average annual soil losses by sheet and rill erosion. The inclusion of the peak storm rainfall rate dependent on the 30 minutes intensity I_{30} (see equation 15) was introduced to account for a greater proportion of soil loss variation from rain storm to rain storm.

The erodibility factor K represents the soil loss rate for a specific soil that is clean-tilled fallow on a 9% slope. This K factor can be derived from the nomograph in Figure 2 of Annex 2. The K factor [kg/cm] converts Q_s values [cm/day] to amounts of erosion [kg/day].

$Q_s \times L_s \times K$ reflects the erosion loss from a continuously fallowed field. Losses from a field with crops and specific erosion control measures are usually much smaller.

The Crop Management Factor CMF is the ratio between the soil loss under specific management conditions and the soil loss if the soil is fallow and intermittently cultivated. A synopsis of some values is presented in Table 6 of Annex 1. If the field is continuously fallow without any management, the CMF is 1. Crops intercept rainfall, reduce puddling of surface soil, help to maintain infiltration rate and slow runoff (Troeh et al., 1980). Crop roots and residues re-ploughed into the soil influence the soil structure, the infiltration rate, the permeability, and therefore lower the erosion loss. As e.g. the vegetation cover of a crop changes during the growing season, one will understand that the CMF actually changes during the season.

The Practice Management Factor PMF indicates the fractional amount of erosion that occurs under specific agricultural conditions in addition to those captured in the CMF. A synopsis of some values is presented in Table 7 of Annex 1. The better the erosion controlling measures, the lower the PMF. The most common of these measures are contour strip cropping, contour cultivation, and terracing with supporting vegetated waterways (Troeh et al., 1980).

$$Q_e = Q_s \times L_s \times K \times CMF \times PMF \quad (equation\ 17)$$

Q_e = *soil loss due to erosion* [$kg.day^{-1}$]
Q_s = *the estimated peak storm rainfall rate (see equation 15)* [$cm.day^{-1}$]
L_s = *the slope-steepness factor (see equation 16)* [-]
K = *the soil erodibility factor (see nomograph in Fig.2)* [$kg.cm^{-1}$]
CMF = *the crop management factor* [-]
PMF = *the practice management factor* [-]

The enrichment ratio r_{om} in equation 18 expresses the increased contribution of smaller particulates to the eroded soil mass. This ratio pertains to the amount of organic matter, as this is — as a rule — the main constituent of this fraction.

$$r_{om} = e^{2 + 0.2 \ln\left(\frac{Q_e}{A}\right)} \quad (equation\ 18)$$

r_{om} = *the enrichment ratio for organic matter* [-]
Q_e = *the soil loss due to erosion* [$kg.day^{-1}$]
A = *area of the watershed* [m^2]

The amount of pesticide that is sorbed to the particulates liable to erosion, can now be estimated by equation 19. The soil water partition coefficient $K_{s/l}$ can be obtained experimentally or empirically (see also equation 9). The term 1000 is introduced to convert milligram to gram.

$$J_e = Q_e \times r_{om} \times K_{s/l} \times \frac{J_r}{V_r} \times 1000 \quad (\textit{equation } 19)$$

J_e = *the substance loss due to erosion* [*g.day*$^{-1}$]

Q_e = *the soil loss due to erosion* (*see equation* 17) [*kg.day*$^{-1}$]

r_{om} = *the enrichment ratio for organic matter* (*see equation* 18) [-]

$K_{s/l}$ = *the soil water partition coefficient* (*see equation* 9) [*dm*^{3}*.kg*$^{-1}$]

J_r = *the estimated actual flux in the subsoil due to runoff* (*see equation* 11) [*g*]

V_r = *the water volume in the subsoil subject to runoff* (*see equation* 12) [*m*^{3}]

In conclusion, the concentration of the pesticide in the adjacent ditch due to erosion of soil particulates by rain storms, can be calculated by equation 20. It should be stressed that this amount of pesticide can be inavailable to organisms in the ditch in case the particulates are not ingested, and in case the pesticide does not desorb from the particulates.

$$C_e = \frac{J_e}{V_d} \quad (\textit{equation } 20)$$

C_e = *the estimated concentration in the receiving water due to erosion* $[mg.l^{-1}]$
J_e = *the estimated actual flux due to erosion* $[g.day^{-1}]$
V_d = *the water volume of the receiving water* $[m^3]$

3.1.4 Overspray

When spraying vehicles have to turn around nearby an adjacent ditch, direct spraying of the surface water by the overhanging boom may occur. The surface water A_o sprayed in this way, is primarily dependent on the number of turns n in a spraying session, and the oversprayed area A_b (equation 21).

$$A_o = n \times A_b \quad (\textit{equation } 21)$$

A_o = *the total oversprayed area* $[m^2]$
n = *the number of turns* [*i.e.* $\frac{W_f}{2W_b}$; W_f *is the width of the field in m,* W_b *is the width of the boom in m*]
A_b = *the oversprayed area per turn* $[m^2]$

The oversprayed area per turn A_b is dependent on the width of the spray boom W_b and the fraction of the boom width β ($0 \leq \beta \leq 1$) that extends over the water (equation 22). The oversprayed area A_b is described with a circle segment.

$$A_b = W_b^2 \times [\arccos(1 - \beta) - ((1 - \beta) \times \sqrt[2]{2\beta - \beta^2})] \quad (\textit{equation } 22)$$

A_b = *the oversprayed area per turn* $[m^2]$
W_b = *the width of the boom* $[m]$
β = *the fraction of* W_b *that extends over the water* [-]

3.2 Dissipation in surface water, after loading

After entering the ditch water, the pesticide may dissipate via advection (equation 23), hydrolysis (equation 24), volatilisation (equation 25), photodegradation (equation 26), sorption (equation 27), and biodegradation outside (equation 28) and inside a WWTP (equation 29).

3.2.1 Advection

$$K_a = \frac{\ln 2}{DT_{50a}} \quad (equation\ 23)$$

K_a = *the advection rate constant* $[d^{-1}]$
DT_{50a} = *the advection half-life* $[d]$

3.2.2 Hydrolysis

$$K_h = \frac{\ln 2}{DT_{50h,\ pH=7}} \quad (equation\ 24)$$

K_h = *the hydrolysis rate constant* $[d^{-1}]$
$DT_{50h,\ pH=7}$ = *the hydrolysis half-life at* $pH=7$ $[d]$

3.2.3 Volatilisation

$$K_v = \frac{\ln 2}{DT_{50v}} \quad (\textit{equation 25})$$

K_v = *the volatilisation rate constant* $[d^{-1}]$
DT_{50v} = *the volatilisation half-life* $[d]$

3.2.4 Photodegradation

$$K_p = \frac{\ln 2}{DT_{50p}} \quad (\textit{equation 26})$$

K_p = *the photodegradation rate constant* $[d^{-1}]$
DT_{50p} = *the photodegradation half-life* $[d]$

It is very difficult to interpret laboratory data on photodegradation, because generally the experiment is carried out with clear liquid. In reality the surface water may be turbid. Therefore, it is advised not to use photodegradation data directly for risk assessment. Sometimes the approximation can be made to equal the DT_{50p} in hours in the laboratory to the DT_{50p} in the field in days.

3.2.5 Sorption

Sorption to sediment is assumed to have the same mechanism and partition coefficient ($K_{s/l}$) as in soil. In case an experimental $K_{s/l}$ is lacking, an empirical may be derived in accordance with equation 9.

$$f_{om} = 1.7 \times f_{oc} \quad \textit{(equation 27)}$$

f_{om} = *organic matter fraction*
f_{oc} = *organic carbon fraction*

3.2.6 Biodegradation

$$K_b = \frac{\ln 2}{DT_{50b}} \quad \textit{(equation 28)}$$

K_b = *the biodegradation rate constant* [d^{-1}] [*excl. WWTP*]
DT_{50b} = *the biodegradation half-life* [*d*] [*excl. WWTP*]

$$K_w = \frac{\ln 2}{DT_{50w}} \quad (equation\ 29)$$

K_w = *the biodegradation rate constant in a WWTP* [d^{-1}]

DT_{50w} = *the biodegradation half-life in a WWTP* [*d*] [*in a WWTP the* DT_{50w} *equals the* $DT_{50,o}$ *in an OECD biodegradability test*]

3.3 Calculations on runoff and erosion

Some calculations on runoff and erosion have been performed to get an idea of the reliability of these modules. Three pesticides were selected showing a gradient in log K_{ow}, water solubility (S), and $K_{s/l}$ values. These were bentazone, ziram[1], and lindane.

	bentazone	ziram	lindane
log K_{ow}	-0.46	?	3.8
S (mg/litre)	500	65	7.3
$K_{s/l}$ (dm^3/kg)	0.02	0.4	32

It could be expected that bentazone would be most liable to runoff, whereas lindane would be most liable to erosion.

Eight scenarios were calculated per pesticide to determine the sensitivity of the module outcomes to differences in the input parameters. The scenarios have been based on extreme differences in input parameters: K is 1, indicating no permeability

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[1] Data on the log K_{ow} of ziram were not available.

of the soil, *versus* K is 0.01, indicating a high permeability; CMF is 1, indicating crop management promoting erosion *versus* CMF is 0.01, indicating crop management preventing erosion as much as possible; I_{30} is 2 cm, indicating a very heavy rainstorm in The Netherlands that roughly occurs once every five years (personal communication KNMI to RIVM, 1995), *versus* I_{30} is 0.02 cm, a much less heavy, and more frequently occurring amount of wet precipitation; RCN is 39, indicating a soil with a low runoff potential, and a high infiltration capacity, *versus* RCN is 94, indicating a soil with a high runoff potential and a low infiltration capacity (these RCNs are the extremes in Table 4). The soil depth over which homogeneous mixing of the pesticide occurs, is assumed to be 0.05 m. The treated field is 100 × 100 m^2, with an elevation of 10 m (i.e. a slope of 10%). A Dutch standard ditch of 100 (length) × 2 (width) × 0.25 (depth) m^3 flows adjacent to the lower side of the field.

The input parameters have been included in Table 1. It should be noted that the outcomes refer to concentrations in the surface water immediately after the rain shower. The outcomes of the calculations with ziram have not been included in Table 1 for reasons of convenience. They show C_r values (i.e. due to runoff) of 4.6 mg/litre, independent of the scenario, and C_e values (i.e. due to erosion) of 7.8×10^{-7} to 12 mg/litre, dependent on the type of scenario. The C_e values for ziram due to erosion were therefore much more close to those for lindane, than to bentazone.

Table 1. Calculations of the concentrations of bentazone (B) and lindane (L) in adjacent surface water due to runoff and erosion. Calculations have been based on eight scenarios per pesticide (B1 - B8 for bentazone, and L1 - L8 for lindane). The dosages X are assumed to be 1.4 and 1.3 kg/ha for bentazone and lindane, respectively. There is only one application per chemical. The concentrations C_r and C_e are in mg/litre.

	B1	B2	B3	B4	B5	B6	B7	B8		L1	L2	L3	L4	L5	L6	L7	L8
I_{30}	2	2	2	2	0.02	0.02	0.02	0.02		2	2	2	2	0.02	0.02	0.02	0.02
RCN	39	39	94	94	39	39	94	94		39	39	94	94	39	39	94	94
WCV	0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35		0.35	0.35	0.35	0.35	0.35	0.35	0.35	0.35
PMF	0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6		0.6	0.6	0.6	0.6	0.6	0.6	0.6	0.6
V_d	50	50	50	50	50	50	50	50		50	50	50	50	50	50	50	50
H	10	10	10	10	10	10	10	10		10	10	10	10	10	10	10	10
L	100	100	100	100	100	100	100	100		100	100	100	100	100	100	100	100
A	10,000	10,000	10,000	10,000	10,000	10,000	10,000	10,000		10,000	10,000	10,000	10,000	10,000	10,000	10,000	10,000
EI	237	237	237	237	59	59	59	59		237	237	237	237	59	59	59	59
R_r	2.4	2.4	2.4	2.4	0.59	0.59	0.59	0.59		2.4	2.4	2.4	2.4	0.59	0.59	0.59	0.59
Q_p	84	84	84	84	21	21	21	21		84	84	84	84	21	21	21	21
S	0.16	0.16	6.4×10^{-3}	6.4×10^{-3}	0.16	0.16	6.4×10^{-3}	6.4×10^{-3}		0.16	0.16	6.4×10^{-3}	6.4×10^{-3}	0.16	0.16	6.4×10^{-3}	6.4×10^{-3}
Q_r	84	84	84	84	21	21	21	21		84	84	84	84	21	21	21	21
$C_{vol,T}$	2.8	2.8	2.8	2.8	2.8	2.8	2.8	2.8		2.6	2.6	2.6	2.6	2.6	2.6	2.6	2.6
$K_{s/l}$	0.02	0.02	0.02	0.02	0.02	0.02	0.02	0.02		32	32	32	32	32	32	32	32

Table 1. CONTINUED

	B1	B2	B3	B4	B5	B6	B7	B8		L1	L2	L3	L4	L5	L6	L7	L8
$C_{vol,W}$	2.7	2.7	2.7	2.7	2.7	2.7	2.7	2.7		0.079	0.079	0.079	0.079	0.079	0.079	0.079	0.079
J_r	2.3×10^6	2.3×10^6	2.3×10^6	2.3×10^6	5.7×10^5	5.7×10^5	5.7×10^5	5.7×10^5		6.6×10^4	6.6×10^4	6.6×10^4	6.6×10^4	1.7×10^4	1.7×10^4	1.7×10^4	1.7×10^4
V_r	8.4×10^5	8.4×10^5	8.4×10^5	8.4×10^5	2.1×10^5	2.1×10^5	2.1×10^5	2.1×10^5		8.4×10^5	8.4×10^5	8.4×10^5	8.4×10^5	2.1×10^5	2.1×10^5	2.1×10^5	2.1×10^5
C_r	2.7	2.7	2.7	2.7	2.7	2.7	2.7	2.7		0.079	0.079	0.079	0.079	0.079	0.079	0.079	0.079
T_c	0.027	0.027	0.027	0.027	0.027	0.027	0.027	0.027		0.027	0.027	0.027	0.027	0.027	0.027	0.027	0.027
Q_s	74	74	74	74	0.74	0.74	0.74	0.74		74	74	74	74	0.74	0.74	0.74	0.74
L_s	2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5		2.5	2.5	2.5	2.5	2.5	2.5	2.5	2.5
K	0.01	0.01	1	1	0.01	0.01	1	1		0.01	0.01	1	1	0.01	0.01	1	1
Q_e	1.1	0.011	111	1.1	0.011	1.1×10^{-4}	1.1	0.011		1.1	0.011	111	1.1	0.011	1.1×10^{-4}	1.1	0.011
r_{om}	1.2	0.48	3.0	1.2	0.48	0.19	1.2	0.48		1.2	0.48	3.0	1.2	0.48	0.19	1.2	0.48
J_e	0.072	2.9×10^{-4}	18	0.072	2.9×10^{-4}	1.1×10^{-6}	0.072	2.9×10^{-4}		3.3	0.013	827	3.3	0.014	5.4×10^{-5}	3.4	0.014
C_e	1.4×10^{-3}	5.8×10^{-6}	0.36	1.4×10^{-3}	5.8×10^{-6}	2.2×10^{-8}	1.4×10^{-3}	5.8×10^{-6}		0.066	2.8×10^{-4}	16	0.066	2.8×10^{-4}	1.1×10^{-6}	0.068	2.8×10^{-4}
C_r+C_e	2.7	2.7	3.1	2.7	2.7	2.7	2.7	2.7		0.14	0.079	16	0.14	0.079	0.079	0.15	0.079

4. DISCUSSION AND CONCLUSIONS

Particularly with respect to the role of runoff and erosion, various formulas and their input data were based on research in the US. Therefore they cannot be translated to European countries without saying. These restrictions should be kept in mind carefully. If one wishes to use this semi-quantitative tool for a specific European region or country, it is useful to consult local experts on e.g. soil science, meteorology, hydrology, and agronomy to adjust the input data in a more realistic way.

Input parameters could be generated by the individual EU countries as this report does not deal with site-specific calculations. It rather provides a framework to prepare further steps for site-specific calculations. It does not e.g. offer default values for each country. Defaults — if needed — have to be established per region or per country by the regulatory authorities.

It should be stressed that the formulas and equations in this report contain various assumptions, simplifications and constraints. This is of course inevitable, if one aims at a semi-quantitative tool which is relatively simple to handle, and need not be run on a computer with elaborate software. Some assumptions are discussed below, rather to give some examples than to present an exhaustive list.

Runoff and erosion are calculated under circumstances that represent a soil with an average uniform vegetation cover. Although it is customarily to relate runoff to the soil moisture resulting from the antecedent rainfall over a 5-day period, here it is assumed that the soil is in an average moisture condition prior to application of the pesticide. Also, interception (the amount of precipitation that remains on leaves, branches, and stems) and water losses as result of evapotranspiration (from foliage) and evaporation (from bare soil) are assumed not to contribute significantly to the amount of water that is available for transport over land. Because it is unlikely that pesticides are applied at below-zero temperatures, where the soil may be concealed by a snow cover, it is assumed here that snow melt due to thawing or rainfall does not contribute to runoff. Also, the spray is not retained by foliage and the total load of the pesticide reaches the soil surface. The pesticide is distributed instantaneously between the soil matrix and moisture; the precipitation starts shortly after the moment of application. One particular problem is that the equations with respect to runoff and water erosion are primarily based on agricultural conditions in the United States, and not on those in Europe.

An example of a simplification by the runoff and erosion module is the amount of soil loss that is expected at a specific length L and elevation H of the watershed. It was shown that convex slopes lose more soil than concave slopes with the same H and L (Troeh et al., 1980). Slopes with a convex top and a concave bottom lose less soil than simple convex or uniform slopes, but faster than simple concave slopes. Another example of a simplification is the Crop Management Factor CMF, as it can be understood that the actual effects of the crop and the type of management are complex and diverse.

What can be concluded from the figures in Table 1? First of all, it has been stated above that the figures have no absolute value; at best they clarify the structure of this semi-quantitative tool, and serve for the purpose of ranking pesticides with a different runoff and erosion *potential*. What about this ranking? The figures show that lipophilic pesticides are roughly 30 times less liable to runoff than hydrophilic pesticides, whereas they are roughly 50 times more liable to erosion. Ziram was expected to be intermediate between bentazone and lindane. However, it resembled much more lindane, than bentazone with respect to the liability to erosion, whereas it showed a higher liability to runoff than bentazone. Therefore the expected ranking has only partially been confirmed.

It can be concluded from Table 1 that with respect to the three pesticides, the concentrations due to runoff are independent of the input parameters. The amount of wet precipitation and the potential infiltration in the soil do not appear to influence the *potential* contribution due to runoff. This is different with respect to erosion: different scenarios indicate differences in concentrations of max. 16×10^6 times. Here the input parameters appear to be much more influential. There appears to be no special dicriminative input parameter causing the most erosion. It is rather the specific combination of parameters that determines the outcomes.

What are the contributions of runoff and erosion — estimated with the module — compared with other routes as e.g. drift, leaching, and atmospheric deposition? This again is a speculative issue, as there are no such data available from field experiments. However, we may draw some conclusions if comparing the module calculations in Table 1 with the outcomes of USES 1.0, an integrated model for risk assessment used by the Dutch (re)registration authorities (RIVM, VROM & WVC, 1994). With this model, concentrations in surface water can be estimated as a function of intrinsic properties as vapour pressure, water solubility, log K_{ow}, and other properties as provided by base set information, e.g. the degradation in standardised laboratory degradation experiments in water/sediment systems. Runoff and erosion are not modelled by USES 1.0. Determining processes are advection, hydrolysis, volatilisation, biodegradation, sedimentation, and resuspension (Linders & Luttik, 1995). The comparison of the estimated initial concentrations is presented below:

Concentration (mg/l)	BENTAZONE	ZIRAM	LINDANE
Initial (C_r+C_e)[1]	2.7	4.6	0.079-16
Initial (USES 1.0)[2]	0.011	0.13	0.00051
Average (USES 1.0)[3]	0.0051	0.0064	2.3×10^{-5}

[1] Figures refer to a 10% slope, and bare soil. The contribution of erosion and runoff are totalled.

[2] Figures refer to the absence of sloping terrain; they refer to Dutch conditions.

[3] Figures refer to the average over one year; they refer to Dutch conditions.

It can be concluded from these figures that the initial concentrations as estimated with USES 1.0 (i.e. assuming one dose per year, and drift percentages of 0.02, 0.1, and 0.001 when using bentazone, ziram, and lindane in maize, fruit, and in arable crops, respectively), are 40 - 31,000 times lower than the initial concentrations as estimated with the module in this report. One may conclude —very provisionally of course — that the *potential* impact of runoff and erosion may exceed the *potential* impact of other sources, as drift, substantially.

Are there other feasible methods to estimate the contributions of runoff and erosion? It was reported that in the U.S. runoff concentrations are calculated as fixed percentages of the applied amount (CUWVO, 1986). Organochlorines were assumed to run off with an amount of 1% of the applied dosage, independent of the type of weather. Granular pesticides were assumed to run off with an amount of 3% of the applied dose, whereas all other pesticides were assumed to show a runoff of 0.5%, unless there had been a very high amount of wet deposition within 14 days after application. If these percentages were used for the three pesticides in Table 1 (bentazone, ziram, and lindane), the concentrations in the adjacent ditch would be $\leq 3.2 \times 10^{-4}$ mg/litre. This is much lower than the calculated outcomes in Table 1.

How realistic are the estimated concentrations in Table 1? No validation has been established in The Netherlands up till now. However, some rough comparisons with concentrations measured in the Dutch field can be made, although a few data are available from the provence of Zuid-Limburg, the only part of the Netherlands where due to the hilly landscape runoff and erosion might be expected to some extent. The max. concentrations of bentazone and ziram in Dutch regional (i.e. no main rivers, and lakes) surface water are 315 and 7.5 μg/litre, respectively (ziram measured as CS_2; RIZA, 1993 and 1995a). Mean concentrations of bentazone nearby treated crops are 0.14 - 6 μg/litre. Concentrations in Zuid-Limburg are not available. These measured concentrations in relatively flat areas are much lower than the estimated total (i.e. due to both runoff and erosion) concentrations in Table 1: 2,700 - 3,100 and 4,600 - 17,000 μg/litre for bentazone and ziram, respectively. Contrarily, lindane has been measured in regional surface water in Zuid-Limburg: 0.00043 - 0.0012 mg/litre. If one assumes that runoff and erosion have contributed primarily to these concentrations — which appears not to be unrealistic beforehand, as monitoring indicates higher concentrations in Zuid-Limburg than elsewhere in The Netherlands (RIZA, 1995b) — one should conclude that the estimated concentrations in Table 1 are far too high (0.079 - 16 mg/litre). However, the measured concentrations refer to the dissolved amount, whereas the estimated concentrations in Table 1 refer to both the dissolved (due to runoff) and the sorbed (due to erosion) amount. A comparison is difficult to make.

In a field trial nearby Hereford (UK) concentrations of lindane have been measured in a small stream, after application at rates of 0.56 kg a.i./ha on an arable field (silty clay loam) with a 6% slope (Williams et al., 1996). The mean concentrations in the stream were 0.000010 - 0.0012 mg a.i./litre, the maximal concentrations were 0.000020 - 0.0045 mg a.i./litre. The peak concentration in this trial (0.0045 mg a.i./litre) was roughly 20 - 4000 times lower than the estimated total concentrations of lindane in a small ditch ($C_e + C_r$ in Table 1). The main route of translocation was probably subsoil runoff (collection via drains). The comparison of these figures indicates that the translocation of lindane with soil particulates is overestimated.

It should be noted that in the Netherlands most slopes are ≤ 2%. However, in the provence Zuid-Limburg arable crops can be grown on slopes up to 18%. Only the use of grassland is permitted on higher slopes. The calculation of C_e and C_r with the same input parameters as in Table 1, except the slope (18% instead of 10%), and except the PMF (0.8 instead of 0.6) reveals that the contribution by runoff remains the same (the amount of runoff is independent of the slope), whereas the contribution by erosion increases six times independent of the scenario. However, the estimated maximal C_e for lindane would exceed the water solubility 13 times, which appears very unrealistic.

One may assume that it is difficult to establish default values in view of the complexity of all dissipation processes. It is interesting to mention that the use of the default values in EPPO (1993) — a 6% slope, 0.2 cm rain/day, 1 kg a.i./ha, K_{ow} is 1000 — in combination with additional assumptions (K 0.5; RCN 60; WCV 0.5; V_d 50 m^3; CMF 0.5; PMF 0.5) result in C_r and C_e values of 0.9 and 0.067 mg/litre. The default value for C_r was reported in EPPO (1993) to be 0.01 mg/litre, which indicates that the runoff part of the module is too conservative.

In conclusion, the value of the module as a first semi-quantitative step is limited. No field data are available that can falsify or confirm the hypotheses put forward by the module. It can be speculated that the module refers to worst case conditions, that may be unrealistic. However, an alternative for this module that combines both requirements of feasibility and validity is not available yet.

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ANNEX 1. Input parameters

WATERSHED COVER VALUES (WCVs), SEE EQUATION 5

Table 2. Watershed Cover Values (WCVs)

SOIL TYPE	WCV VALUES		
	CULTIVATED	PASTURE	WOODLANDS
above average infiltration; sandy or gravely	0.20	0.15	0.10
average infiltration rates; loams and such	0.40	0.35	0.30
below average infiltration rates; heavy clay soils; clay pans	0.50	0.45	0.40

RUNOFF CURVE NUMBERS (RCNs), SEE EQUATION 6

Agricultural soils can be classified according to general hydrologic properties (McCuen, 1982)[2]. Four major groups of soil types can be indicated independently of watershed slope and cover (see Table 3):

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[2] McCuen, R.H. (1982) A guide to hydrologic analysis using SCS methods. Prentice Hall Inc., Englewood Cliffs, New Jersey.

Table 3. Major groups of soil types.

A	Low runoff potentials; high infiltration rates (0.76 - 1.14 cm/hr) even when thoroughly wetted; deep and excessive drainage; high rates of water transmission; gravel and coarse sand, deep sand, deep loess, aggregated silts.
B	Moderate infiltration rates (0.38 - 0.76 cm/hr) when thoroughly wetted; moderately to well drained; moderate to fine coarse structures; sandy loam, shallow loess.
C	Slow infiltration rates (0.13 - 0.38 cm/hr) when thoroughly wetted; layers that impede downward movement of water; moderately fine to fine textures; clay loams, shallow sandy loam; low organic content.
D	High runoff potential; slow rates of water transmission; very slow infiltration rates (0.03 - 0.13 cm/hr) when thorougly wetted; soils with a high permanent water table; soils with clay pans or clay layers near the surface; shallow soils over impervious materials; clay soils with a high swelling potential.

These soil groups can be combined with different types of (vegetative) cover (see Table 4).

Table 4. Hydrologic condition of some agricultural areas

PASTURE OR RANGE	
Hydrologic condition	*Vegetative condition*
Poor	Heavily grazed; no mulch; plant cover less than c. 50% of the area.
Fair	Moderately grazed; between 50 and 75% of the area with plant cover.
Good	Lightly or not grazed; more than 75% of the area with plant cover.
WOODLANDS	
Poor	Heavily grazed or regularly burned; no litter, small trees or brush.
Fair	Not fully protected against grazing; not burned; some litter.

With this classification RCN values can be given for a number of agricultural covers. In the following table only values for straight row crops are given. RCNs for contoured and terraced fields can be found in textbooks, e.g. Ven Te Chow (1964)[3], or in publications from the US Soil Conservation Service.

Table 5. Runoff Curve Numbers (RCNs) for hydrological soil cover complexes

		RCN FOR SOIL GROUP			
LAND USE	HYDROLOGIC CONDITION	A	B	C	D
FALLOW	Poor	77	86	91	94
ROW CROPS	Poor	72	81	88	91
	Good	67	78	85	89
SMALL GRAIN	Poor	65	76	84	88
	Good	63	75	83	87
CLOSE-SEEDED LEGUMES	Poor	66	77	85	89
	Good	58	72	81	85
PASTURE OR RANGE	Poor	68	79	86	84
	Fair	49	69	79	80
	Good	39	61	74	88
WOODLANDS	Poor	45	66	77	83
	Fair	36	60	73	79
	Good	25	55	70	77

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[3] Ven Te Chow (ed) (1964) Handbook of applied hydrology. MacGraw Hill, New York.

ERODIBILITY FACTOR (K), SEE EQUATION 17

The erodibility factor (K) converts units of Q_s to amounts of erosion ($0 < K \leq 1$; Troeh et al., 1980)[4]. K represents the soil loss rate for a specific soil as measured on a unit plot, which is defined as a 21.8 m length of uniform 9% slope in clean-tilled fallow. A nomograph was developed relating erodibility to easily measured soil properties (see Fig.2). These properties include the percentage of silt and very fine sand (2 - 100 μm), the percentage organic matter (0 - 4%), the percentage sand (100 - 2000 μm), the Soil Structure Code (SSC; see below), and the Permeability Code (PC; see below). The SSC indicates whether the soil structure is very fine granular (SSC is 1), fine granular (SSC is 2), medium or coarse granular (SSC is 3) or blocky, platy, or massive (SSC is 4). The PC indicates whether the permeability is rapid (PC is 1), moderate to rapid (PC is 2), moderate (PC is 3), slow to moderate (PC is 4), slow (PC is 5), or very slow (PC is 6).

The dotted line in Fig.2 shows how the nomograph is used to obtain a K value of 0.41 for a soil having 65% silt + very fine sand, 5% sand, 2.8% organic matter, fine granular structure, and slow to moderate permeability. The same sequence of properties must always be used to obtain K values from the nomograph (after Troeh et al., 1980)[5]

CROP MANAGEMENT FACTOR (CMF), SEE EQUATION 17

The Crop Management Factor is the ratio between the soil loss under specific management conditions and the soil loss if the soil is fallow and intermittently cultivated ($0 < CMF \leq 1$; Troeh et al., 1980)[5]. The soil loss from a field that is continuously fallow is the product of Q_s, L_s and K (see equation 17). CMFs are extensively reported in various sources (e.g. Troeh et al., 1980[5]; Carsel et al., 1984[6]). For illustrative purposes only, a few selected values from these sources for moderate productivity levels (unless stated otherwise) are presented (see Table 6). If a field is continuously fallow without any management the CMF equals one.

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[4] Troeh, F.R., J.A. Hobbs & R.L. Donahue, eds. (1980) Soil and water conservation for productivity and environmental protection. Prentice-Hall Inc., Englewood Cliffs, New Jersey.

[5] Troeh, F.R., J.A. Hobbs & R.L. Donahue, eds. (1980) Soil and water conservation for productivity and environmental protection. Prentice-Hall Inc., Englewood Cliffs, New Jersey.

[6] Carsel, R.F. et al. (1984) User's manual for the Pesticide Root Zone Model (PRZM). Environ. Res. Lab. Office of Research and Development. EPA, Athens, Georgia.

Figure 2. A nomograph to determine the soil-erodibility factor K (after Troeh et al., 1980)

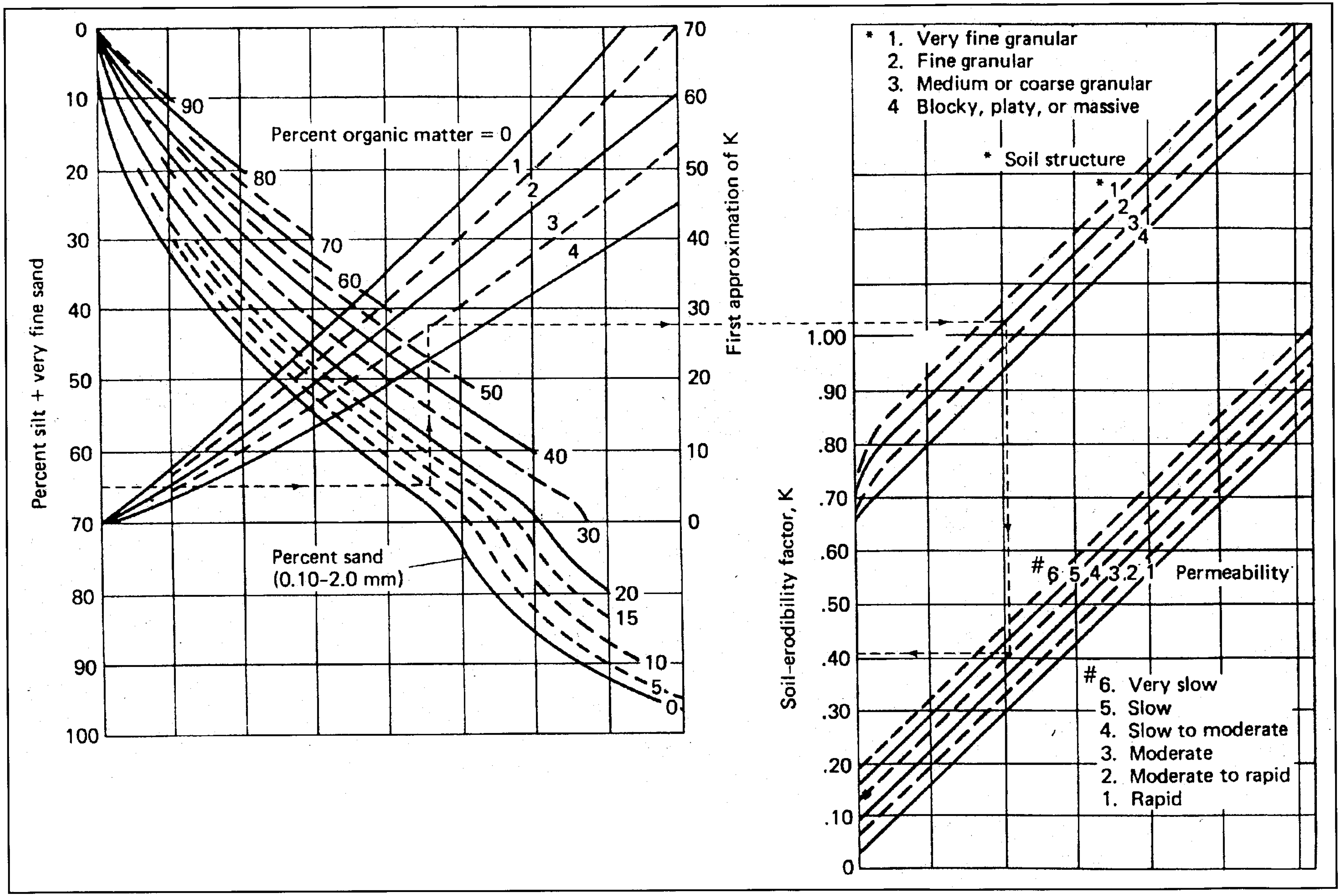

Table 6. Crop Management Factors (CMF) for some agricultural management types.

CROP, ROTATION AND MANAGEMENT	CMF
Corn; residues removed; turn plowed in fall; no rotation	0.62
Corn; shredded stalks; turn plowed in spring; no rotation	0.44
Corn; not tilled; planted in chemically killed sodds; 80% of surface is covered with mulch; no rotation	0.053
Corn-wheat-grass rotation (3 year cycle); all residues left on field; turn plowed for corn; disk plowed for wheat	0.095
Meadow; grass	0.01
Meadow; sweet clover; high productivity	0.025
Wheat-fallow rotation (2 year cycle); turn plowed after wheat; high productivity	0.38
Winter wheat; all residues left on field; turn plowed in August; conventional management; no rotation	0.19
Wheat-grass rotation (2 year cycle); conventional management; high production	0.054

PRACTICE MANAGEMENT FACTOR (PMF), SEE EQUATION 17

The Practice Management Factor indicates the fractional amount of erosion that occurs with specific agricultural practices in addition to those captured in the CMF. Examples are contour cultivation, terracing with supporting vegetated waterways and contour strip cropping. Some examples of PMFs are presented in Table 7.

Table 7. Practice Management Factors (PMFs) for contouring on various slopes (after Troeh et al., 1980[7]).

LAND SLOPES	PMF
1 - 2%	0.60
3 - 5%	0.50
6 - 8%	0.50
9 - 12%	0.60
13 - 16%	0.70
17 - 20%	0.80
21 - 25%	0.90

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[7] Troeh, F.R., J.A. Hobbs & R.L. Donahue, eds. (1980) Soil and water conservation for productivity and environmental protection. Prentice-Hall Inc., Englewood Cliffs, New Jersey.

ANNEX 2. Emission factors used for drift in the Netherlands [8]

Location and type of application		F_d
1. *Indoor applications*[9] (excl. greenhouses)	- storage cells, etc.	0
	- shower rooms, etc.	0
2. *Greenhouse applications*		
a. Specific applications	- overhead irrigation	0
	- manual pouring	0
	- soil treatment	0
	- granule application	0
	- trickling	0
	- chicory pit for silage	0
b. Non-specific applications	- remaining types of application in greenhouses (spraying, mist blowing, fogging, smoke generating, etc.): emission mainly through condensation on glass roof[10]	0.001
3. *Field applications*		
a. Specific applications	- manual pouring	0
	- dipping	0
	- granule application[11]	0
	- baiting	0
	- soil/plant injection	0
	- plant base treatment	0
	- smearing	0
	- brushing	0
	- spraying with direct incorporation into soil[12]	0
	- seed treatment	0
	- roof shielding	0
b. Spot applications	- waste dump (e.g. of potatoes)	0.005
	- row spraying[13]	0.005
	- knapsack spraying	0.005
	- road signs	0.005

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[8] A list updated with emission factors for drift is in the making (dec 1996).

[9] Whenever no direct exposure of surface water by drift is to be expected through the type of application, the load via this route is determined to be 0%.

[10] From research on condensate discharge it was derived that approximately 0.1% of the plant protection product dosage on the glass roof can load to the surface water via condensate. Up to now, it has been impossible to specify per type of application.

[11] With a special synthesis-granulate spreading device.

[12] Spraying with direct incorporation into the soil during a single run.

[13] This figure is based on the assumption that during row spraying less drift will occur than during field application as the distance from nozzle to soil is substantially less during row spraying than during full field treatment.

ANNEX 2 CONTINUED

c. Non-specific applications	1. *crop height ≤ 25 cm*[14]:	
	- soil treatment	0.01
	- bare soil	0.01
	- herbicide in fruit culture	0.01
	- directed spraying of soil below crop	0.01
	- plant/seed bed	0.01
	- before germination	0.01
	- asphalted terrain, pavements	0.01
	2. *crop height > 25 cm*:	
	- downward spraying	0.02
	- treatment of field bordering ditches	0.05
	- treatment field border	0.05
	- sideways or upward-directed spraying in arboriculture and fruit-growing	0.1
	3. *ditch slope application*	0.1
d. Specific applications	- spraying by aircraft	1
	- willow-coppice	1
	- dry ditch sediment	1

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[14] During applications to bare soil and (still) low crops, a relatively low emission is assumed because the spraying arm could be adjusted closer above the crop.

ANNEX 3. Glossary

advection	intercompartmental transfer of a chemical by a carrier that physically flows from one compartment to the other; advective mass transfer is strictly one-way; examples are atmospheric deposition, sedimentation, resuspension
CMF	the Crop Management Factor is the ratio between the soil loss under specific management conditions and the soil loss if the soil is fallow and intermittently cultivated ($0 \leq CMF \leq 1$)
DT_{50a}	the advection half-life
DT_{50b}	the biodegradation half-life
DT_{50h}	the hydrolysis half-life
DT_{50o}	the biodegradation half-life in an OECD biodegradability test
DT_{50p}	the photodegradation half-life
DT_{50v}	the volatilisation half-life
DT_{50w}	the biodegradation half-life in a WWTP
K	the erodibility factor K indicates the soil loss rate for a specific clean-tilled fallow soil on a 9% slope. This factor can be derived from a nomograph ($0 \leq K \leq 1$)
K_a	the advection rate constant
K_h	the hydrolysis rate constant
K_p	the photodegradation rate constant
K_v	the volatilisation rate constant
K_{om}	the sorption coefficient normalised to the fraction of organic matter in soil

K_{ow}	octanol-water partition coefficient
$K_{s/l}$	a soil-water partition coefficient —or sorption coefficient — *independent* on the ratio 1/n (n is an empirical entity which describes the non-linearity of an adsorption isotherm)
PC	the Permeability Code indicates the permeability rate of the soil (PC is 1, 2, 3, 4, 5, or 6). The PC is used for determining the K factor in a nomograph
PMF	the Practice Management Factor indicates the fractional amount of erosion that occurs with specific agricultural practices in addition to the those captured in the CMF
RCN	the Runoff Curve Number indicates the potential runoff as a function of general hydrological conditions (e.g. infiltration rate), the extent of grazing, and the type of land use
SSC	the Soil Structure Code indicates the texture of a soil from very fine granular to blocky, platy or massive (SSC is 1, 2, 3, or 4). The SSC is used for determining the K factor in a nomograph
WCV	the Watershed Cover Value indicates the average infiltration rate in relation to the vegetation cover of a watershed ($0 \leq WCV \leq 1$)
WWTP	Waste Water Treatment Plant