Simulation of phosphate leaching in catchments with phosphate-saturated soils in the Netherlands

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ABSTRACT

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The effects on phosphate leaching to surface waters of two scenarios for net phosphate input to sandy agricultural soils were estimated. WATBAL and ANIMO simulations for manure surplus areas in the Netherlands were used. The methodology and models were verified by comparing model results with measured values of the Schuitenbeek catchment. Simulated values of phosphate loads to surface waters and phosphate concentrations were underestimated by 10 and 30% respectively. Phosphate leaching to surface waters and phosphate concentrations hardly increase in the lower scenario with a net P_2O_5 input or 'loss' of 10 kg ha^{-1} a^{-1}. With a P_2O_5 surplus of 40 kg ha^{-1} a^{-1} both leaching fluxes and concentrations increase.

Keywords: manure surplus, overfertilization, simulation model, surface water quality

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Preface

In this report an overview is presented of the results of a study by DLO - Winand Staring Centre carried between 1993 and 1995 in which the phosphate leaching from areas with a manure surplus was simulated as a function of future fertilization scenarios. This study was a follow up of the studies by Reijerink and Breeuwsma (1993) and Reijerink et al. (1993) who investigated the degree of phosphate saturation and the location of phosphate saturated soils in these areas.

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Summary

Because of the very high livestock densities in some regions in the Netherlands, phosphate loads to agricultural land by application of manure lead to substantial phosphate surpluses, especially on sandy soils that have a small buffer capacity for phosphate. As a result, present phosphate leaching in these area exceeds threshold values for surface water quality and adverse effects on the surface water quality occur. To investigate whether the phosphate leaching can be decreased by limiting the allowed P surplus, a modelling study was carried out to evaluate the effects of two different fertilization scenario's. In the first scenario, a phosphate surplus of 10 kg P_2O_5 was used whereas in the second scenario a surplus of 40 kg P_2O_5 was used. To estimate the phosphate leaching from soils in the area with sandy soils and a manure surplus in the Netherlands, simulations were carried out with dynamic models for a scenario's with a phosphate surplus of 10 kg . Computations were made for about 2100 units unique in soil chemistry, soil physics, hydrology, historic phosphate load and land use. The hydrology was simulated with the simple, two layer dynamic model WATBAL. Nutrient related processes were simulated with the model ANIMO. After initialization of the models with the historic phosphate loads, two different scenario's were run for a period of 60 years.

The methodology and models were verified on the Schuitenbeek catchment. In this catchment, measurements of phosphate leaching and phosphate concentrations were available for the period 1990-1993 which allow a comparison of simulated with measured values. Results of the verification show that the simulated phosphate saturation as a function of soil depth was within the range of measure values. Results, however, also indicate that the simulated phosphate front is sharper than the measured one which causes leaching and concentrations of phosphate that are somewhat lower than the measured data, especially for groundwater regime class V/V*. The comparison was hampered by the fact that the hydrologic years 1990-1993 were not in the meteorological data set used to compute the hydrology.

The effects of the scenario's was evaluated on some selected plots. From these plot calculations it can be concluded that leaching at the level of the average highest ground water level (GHG) tends to the level of the added surplus. Leaching fluxes towards surface waters increase with the +40 kg scenario, with the +10 kg scenario leaching fluxes are stabilized.

A first tentative assessment was made of the effects of a chemical treatment to reduce the leaching of phosphate, by assuming that this treatment was applied on all strongly P saturated soils in each catchment and that this treatment reduces P leaching by 70%. Results show that such a measure can only reduce P leaching on the catchment level if a scenario with a low P surplus for the other soils (with a P saturation < 75%) is used. In that case a reduction of maximal 20-30% in phosphate leaching was calculated. Higher reduction percentages are possibly only feasible in small (sub) catchments. In case of a larger surplus (40 kg), the positive effects of a chemical

treatment for very strongly P saturated soils are diminished by an increased P leaching from other soils.

Results of the scenario runs for the entire sandy area show that almost all evaluated units, the units with groundwater regime classes I up to V*, have a phosphate saturation higher than 25% which means that these units are phosphate saturated. This is in accordance with results from Reijerink and Breeuwsma (1993).
Phosphate leaching and phosphate concentration for the +10 kg scenario hardly increase between year 20 and year 50 of the scenario run. For the +40 kg scenario, however, an increase in time for both concentration and leaching is simulated.

Samenvatting

Simulatie van fosfaatuitspoeling in stroomgebieden met fosfaatverzadigde gronden in Nederland

Door de sterke toename van intensieve veehouderij sinds 1970 in het oostelijk-, centraal- en zuidelijk zandgebied, zijn er in deze gebieden grote fosfaatoverschotten onstaan. Als gevolg hiervan zijn veel van de gronden in deze gebieden fosfaatverzadigd geraakt met als gevolg een toename in de fosfaatuitspoeling en negatieve effecten op de kwaliteit van het oppervlakte water.

Aansluitend op eerder onderzoek naar de fosfaatverzadiging van de bodem in mestoverschotgebieden is nu door DLO - Staring Centrum de te verwachten fosfaatuitspoeling naar grond- en oppervlaktewater berekend voor verschillende verliesnormen teneinde na te gaan of een verlaging van de verliesnorm leidt tot een vermindering van de fosfaatuitspoeling op de midellange termijn (± 50 jaar). Tevens is globaal nagegaan wat de invloed van een sanering van de sterkst verzadigde gronden is op de fosfaatuitspoeling. Het onderzoek werd in de periode 1993-1995 uitgevoerd in opdracht van het Ministerie VROM.

Bij het onderzoek werd gebruik gemaakt van het gegevensbestand van de fosfaatverzadigde gronden, het nutriëntenmodel ANIMO en het hydrologische model WATBAL. De berekeningen werden uitgevoerd voor stroomgebieden die een onderdeel vormen van de PAWN-districten (afkomstig van VROM, afd. Emissieregistratie). Kwelgegevens zijn door RIZA berekend met het model NAGROM. De berekeningen zijn getoetst via meetgegevens van het RIZA voor het Schuitenbeekgebied. Daarbij bleek dat het gesimuleerde fosfaatfront in de bodem iets scherper is dan in de praktijk wordt waargenomen. De berekende fosfaatbelasting van het oppervlaktewater is ongeveer 10% lager dan de meetwaarde en de fosfaatconcentratie 30%. De lichte onderschatting wordt waarschijnlijk veroorzaakt door het scherpere fosfaatfront en een lagere GHG (gemiddeld hoogste grondwaterstand) bij de modelberekeningen.

De uitspoeling naar het grondwater is alleen nagegaan voor enkele karakteristieke combinaties van gewas, bodem, grondwatertrap en fosfaatverzadigingsgraad. Deze uitspoeling is na verloop van enkele tientallen jaren gelijk aan het toegediende overschot (de verliesnorm). In de eerste jaren na invoering van de verliesnormen kan de uitspoeling naar het grondwater zowel toe- als afnemen, afhankelijk van de hoogte van de verliesnorm en de fosfaatverzadiging.

Doordat in de ondergrond nog fosfaat wordt vastgelegd is de uitspoeling naar het oppervlaktewater aanzienlijk lager. Zelfs bij een lage verliesnorm van 10 kg P_2O_5 per ha is de uitspoeling naar het oppervlaktewater echter nog een factor 1-6 hoger dan overeenkomt met de norm (grenswaarde) voor de oppervlaktewaterkwaliteit. De belasting van het oppervlaktewater neemt bij deze lage verliesnorm niet verder toe doordat het fosfaatoverschot volledig in de ondergrond wordt vastgelegd. Bij een verliesnorm van 40 kg P_2O_5 per ha is dit niet langer het geval. Gemiddeld over

alle stroomgebieden bedraagt de fosfaatbelasting van het oppervlaktewater na 50 jaar bij de genoemde verliesnormen 2,9 resp. 3,3 kg P_2O_5 per ha per jaar. De variatie tussen stroomgebieden bedraagt ongeveer 1-6 kg P_2O_5 per ha per jaar.

Sanering van de sterkst verzadigde gronden, met een fosfaatverzadigingsgraad van meer dan 75%, leidt op stroomgebiedniveau ook bij een lage verliesnorm (10 kg P_2O_5 per ha) slechts tot een beperkte reductie van de uitspoeling van maximaal 20 à 30%. Hogere reductiepercentages zijn waarschijnlijk alleen bij zeer kleine stroomgebieden te realiseren.

Het aantal gebieden waar sanering effect heeft daalt sterk bij een hogere verliesnorm van 40 kg P_2O_5 per ha.

1 Introduction

Since 1970, livestock densities have increased substantially in sandy districts in the central, eastern and southern part of the Netherlands. As a result, application of phosphate in manure commonly exceeded crop requirements and caused phosphate saturation as shown in previous studies (Breeuwsma and Schoumans, 1987, Breeuwsma et al., 1990, Reijerink and Breeuwsma, 1993). The potential impact of phosphate saturation on leaching has since long been recognized (Lexmond et al., 1982). An example of present impacts was given by a field study in the catchment area of the Schuitenbeek (Breeuwsma et al.,1989) and a modelling study in the same area (Schoumans and Kruijne, 1995). However, an overall picture of the present and future impact of phosphate saturation in the sandy districts on the loads to surface waters was not available. This knowledge is of great importance to agricultural and environmental policies and to (regional) water authorities. The present study was cofinanced by the Ministry of Housing, Physical Planning and Environment.

The primary objective of this study was to quantify the impact of phosphate saturation on the leaching of phosphate to surface waters for all catchments in the sandy districts in central and southern Netherlands as a function of time and application rates. The areas investigated are indicated on Map 1 of chapter 3. Calculations of phosphate concentration and leaching are based on data collected in previous studies (e.g. Reijerink et al., 1993). The ANIMO model version 3.5 (Kroes, in prep.) is used to describe the fate of phosphate in soils. This version of ANIMO contains the detailed phosphate module described by Schoumans (1995). The transport of water was calculated with the hydrological model WATBAL (Berghuys-Van Dijk, 1985).

Chapter 2 of this report contains a short description of model formulations, required input and procedures for initialization and calibration. The input data on soils, ground water, land use and historical phosphate loads are discussed in Chapter 3. Chapter 4 describes the results of a model validation in the catchment area of the Schuitenbeek. Results of the model application on the sandy districts are described in Chapter 5. The leaching from the catchments is presented on maps for two future scenario's of P inputs. The long term effects are illustrated for various combinations of soils, ground water regime and historical loads, using plots of leaching versus time. Chapter 6 mentions the conclusions and recommendations.

2 Methodology

To model phosphate leaching towards surface waters two simulation models were used: the hydrological model WATBAL and the water quality model ANIMO, in which solute transport and the biogeochemical reactions of the nutrients C, N and P are simulated. The hydrology simulated with the model WATBAL was used as input for ANIMO.

2.2 Hydrology

2.2.1 Model

For the hydrological modelling the model WATBAL (WAter BALance) was used. WATBAL (Berghuijs-Van Dijk, 1985) is a simple dynamic water balance model. The soil is divided into two layers i.e. the rootzone and a layer underneath it, the subsoil. The model calculates for each timestep the changes in water content of the two layers. The position of the phreatic surface follows from the calculated water contents. The following processes are included in the model WATBAL: precipitation, evapotranspiration, capillary rise, percolation, drainage to a maximum of 4 drainage systems (such as canals, ditches and trenches) and surface runoff. Important input data are soil physical parameters, drainage base, drainage resistance, crop factors, soil cover data, precipitation and open water evaporation.
The concept of the model is based on the theory of Ernst (1962,1978) on the stationary flow in the saturated zone in the occurrence of parallel drains. Fluxes towards drainage systems are calculated according to:

$$q_d = \frac{h - hd_i}{c_i} \qquad (1)$$

in which:

q_d	= flux towards drainage system i	(m d^{-1})
h	= water level	(m)
hd_i	= drainage depth of drainage system i	(m)
γ_i	= drainage resistance of drainage system i	(d^{-1})

Drainage and seepage at the bottom boundary are calculated according to formula (1) in which hd_i and γ_i are used to achieve a certain drainage or upward seepage flux. Because the seepage flux is not constant throughout the year, hd_i is given as a sinus function. With the amplitude of the sinus function the difference between maximum and minimum seepage is set. With the phase shift the day at which the seepage is at it's maximum can be set.

The flux from the rootzone towards the subsoil is calculated as:

$$q_t = \frac{M - M_0}{M_s - M_0} * K_{sat} \tag{2}$$

in which:

q_t	= flux from rootzone to subsoil	(m d^{-1})
M	= actual amount of water	(m)
M_0	= amount of moisture at field capacity	(m)
M_s	= amount of moisture at saturation	
K_{sat}	= saturated hydraulic conductivity	(m d^{-1})

Actual evapotranspiration is calculated from the open water evaporation with a reduction function as described by Rijtema and Aboukhaled (1975).

2.2.2 Input for WATBAL

For this study meteorological data, precipitation and open water evaporation from Meteorological Station de Bilt were used for the period 1953-1967. This series of data was repeated to allow for longer model calculations. Because there is not a data set on drainage depths and drainage resistances which covers the whole study area , drainage depths and drainage resistances were derived from model calibrations for two different crops and five different ground water regime classes.
Soil physical parameters were derived from the 'Staringreeks' (Wösten et al., 1994), which gives average soil physical characteristics for the most important Dutch soil texture classes. Used characteristics are *h-theta* relations, values for *theta* at saturation, field capacity and wilting point, and the height of capillary rise. Seepage fluxes at the bottom boundary of the soil profile are treated as model input and were calculated at RIZA with the groundwater model NAGROM (NAtional GROundwater Model) (De Lange, 1991). Seepage fluxes were clustered into five different seepage classes. Table 7 in paragraph 3.5 gives the seepage fluxes for each seepage class.

2.2.3 Model calibration

Values for drainage resistances and the depths of drainage systems were derived by calibration of the model WATBAL. Calibrations were carried out for three different crops, in combination with six different soil physical units in combination with five different ground water regime classes for five seepage classes. The parameters (drainage level and drainage resistance) were optimized to fit a frequency distribution of ground water levels specific for a certain ground water class. Optimization was done by searching the parameter set giving the best agreement between the simulated frequency distribution and the distribution as derived by Van der Sluys. To that end

WATBAL was coupled to the IMSL routine ZXSSQ (IMSL, 1982), which uses a Levenberg Maquardt iteration scheme to find the set with the lowest sum of squares.

Ground water regime classes in the Netherlands are classified according to their average highest water level (GHG) and their average lowest water level (GLG). These values are calculated as the average of the measured three lowest and three highest water levels within a year from two monthly measurements around day 14 and day 28 over a long time span (10 to 20 years) . Van der Sluys (1982) derived a continuous function for the frequency distribution of exceedances of ground water levels:

$$Y(P)=b_0+b_1X_1+b_2X_2 \tag{3}$$

with:

$Y(P)$	=	ground water level exceeded during P% of the time	(-)
X_1	=	GHG	(m)
X_2	=	GLG	(m)

the coefficients b_0, b_1 and b_2 are a function of P. For [5<P<95] the coefficients are:

$$b_0=8.9+0.123P-1.7127.10^{-2}P^2+1.5348.10^{-4}P^3 \tag{4}$$

$$b_1=0.97+1.591.10^{-3}P-2.2765.10^{-4}P^2+1.01967.10^{-6}P^3 \tag{5}$$

$$b_2=-0.23+1.4732.10^{-2}P \tag{6}$$

For this study five aggregated ground water regime classes were used. An overview of the classes with associated GHG and GLG are listed in Table 5 in Section 3.2. The ground water regime classes VI and higher were not used in this study because soils with these low water table hardly contribute to phosphate leaching towards surface waters (Schoumans and Kruijne, 1995).

Some of the soil physical units which have almost similar soil physical properties, were clustered to minimize the number of calibrations and computations.

Calibration was performed for maximal three different drainage levels. The drainage levels are: level 1: interflow, level 2: ditches, level 3: deep ditches/canals. Table 1 gives the drainage levels used for each ground water regime class (GWC).

Table 1 Drainage levels used for calibration

GWC	Level 1	Level 2	Level 3
1	+	+	+
2	+	+	+
3	-	+	+
4	+	+	+
5	-	+	+

To minimize the degrees of freedom, not all parameters were optimized and some had a preset value. The depth of the drainage level 1 which accounts for interflow and surface runoff was set to 20 cm below the soil surface for all groundwater regime classes. The depth of the second drainage level was set to 10 cm below the average spring water level (GVG). GVG is calculated from GHG and GLG according to (Van der Sluijs, 1982):

$$GVG = 5.4 + 1.02\,GHG + 0.19\,(GLG - GHG) \tag{7}$$

with: GVG = average spring water level (m)

In general the calibration method used in the optimization worked well , although for some combinations there were convergence problems. Furthermore calibration did not always lead to a unique set of drainage parameters. In such cases the ultimate values for the parameters depended on the values given to the parameters at the start of the calibration. However calculations with ANIMO using hydrology computed with these different parameter sets showed very small differences in leaching fluxes.

2.3 Phosphate leaching

2.3.1 Model description

This paragraph gives a short description of the water quality model used. The description is focused on phosphate.

To model phosphate leaching towards surface- and ground water, the model ANIMO was used (Rijtema, 1993). The water quality model ANIMO (Agricultural Nitrogen Model) is a dynamic simulation model that simulates the carbon, nitrogen en phosphate cycle and their interactions. The model uses a one dimensional soil profile, which is divided in a number of horizontal layers and it calculates lateral fluxes to and from surface waters. A more detailed description of the model can be found in Kroes (in prep). Processes included in ANIMO which influence the transport of phosphate through the soil profile and towards surface waters are:
- solute transport (vertical and lateral)
- addition of manure to the soil
- plant uptake of nutrients
- turnover of plant residues

- mineralisation and immobilisation of organic matter
- sorption and desorption of phosphate by the soil matrix
- precipitation and dissolution of phosphate salts

Water transport as calculated by the model WATBAL was used as input for ANIMO. From the water fluxes within the soil profile and the fluxes to and from the drainage systems solute transport of dissolved matter is calculated.

To simulate solute transport the one dimensional vertical soil profile is divided in horizontal layers. Via lateral fluxes evapotranspiration and discharge towards drains is simulated. Furthermore, vertical transport from layer to layer is simulated. Drainage to deep soil water and seepage are simulated as vertical transport. For each timestep a complete water and mass balance is solved for each layer.

For the transport to drainage systems, 3 drainage levels can be used in ANIMO with their own drainage resistance and drainage base. Each timestep the thickness of the discharge layers to the different drainage systems is calculated. The thickness of the discharge layer per drainage system is proportional to the discharge for each drainage system. Discharge takes place between the soil water level and the bottom of the soil profile.

The ANIMO model accounts for different kinds of manure each with its own composition. In this study simulations were carried out using pig slurry only. The manure is divided in a mineral part and an organic part, the organic part is divided over different fractions with their own solubility, P and N content and mineralization rate.

The addition of manure takes place on the soil surface or directly into the soil. In this study all manure was added to the soil surface only. The manure mixes with the precipitation and is transported into the soil. Ploughing of the soil was not considered.

For grassland plant uptake is a function of the nutrient status of the soil. For maize a maximum uptake is defined. Uptake for both crops is limited by the actual evapotranspiration in a year.

Organic matter in the soil is composed of fresh organic material and humic material. Plant residues from roots, harvest losses and exudates of maize roots are additional sources of organic matter in soil. Each material has it's own composition and specific mineralization rate.

Because sorption and desorption reactions are very important for phosphate leaching a short description of reaction mechanisms and process descriptions in ANIMO (version 3.5) is given in this paragraph. Schoumans (1995) gives a comprehensive description of the reaction mechanisms for inorganic phosphate reactions, and the values for the model parameters using laboratory data. ANIMO version 3.5 has a new phosphate sorption module based on this description (Groenendijk et al., in prep.). The module has been validated at the field scale (Kruijne et al., 1995; Schoumans and Kruijne, 1995) and a regional scale (Schoumans and Kruijne, 1995).

Two reaction mechanisms are distinguished for the sorption of phosphate in soils. The first is a sorption reaction at the surface of Al- and Fe- (hydr)oxides, the second may be visualized as a diffusion reaction into the aggregates of these Al- and Fe-(hydr)oxides.

The surface reaction is a fast reaction, with reaction times in the order of hours to days. This sorption process is assumed to be completely reversible. The surface reaction is in equilibrium with the phosphate concentration in solution. In ANIMO three options are available to describe this reaction: a linear sorption model, a Langmuir model or a Freundlich model. According to Schoumans (1995), and in line with the protocol for phosphate saturated soils (Van der Zee, 1990) we used the Langmuir model. Annex 1 gives sorption equations and parameter values.

The diffusion reaction is a slow process, and the diffusion rate slows down with increasing amounts of diffused phosphate. It is very hard for already diffused phosphate to come into solution again. This is possible only after almost all sorbed phosphate is desorbed. Therefore in practice the diffusion of phosphate in aggregates of Al- and Fe- oxides can be regarded as an irreversible process. The model ANIMO has three options for the diffusion reaction: a time dependent linear sorption model, a time dependent Langmuir model or a time dependent Freundlich model. According to Schoumans (1995) we used a summation of three Freundlich equations to describe the slow reaction. The sum of these three equations results in an S-shaped curve as found by Van der Zee (1990).

Precipitation of phosphate salts occurs if the phosphate concentration in solution rises to the buffer concentration. The soil matrix is then completely occupied with sorbed phosphate. The buffer concentration for the pig slurry used in this study is assumed to be 50 $mg.l^{-1}$ P. Precipitation of phosphate salts is modelled as an instantaneous reversible process.

2.3.2 Model input

Model input for ANIMO can be divided into 8 groups: properties of (organic) materials, data for crop growth and crop uptake, soil parameters, boundary conditions, initial conditions, chemical parameters, manure addition and hydrological input.

Definition of the organic materials is according to Schoumans and Kruijne (1995) for the calculation of phosphate leaching towards surface waters in the catchment of the brook 'Schuitenbeek'.

Crop growth parameters were taken from the 'Cranendonck' dataset (Kroes, 1994), except for the uptake parameters for grassland which were calibrated to simulate an average net uptake of 100 kg P_2O_5 for grass.

The soil parameters and phosphate applications used in the simulations are described in chapter 3.

Boundary conditions such as the concentration of P and N in seepage water and atmospheric deposition of N and P were taken from the 'Cranendonck' dataset and were kept constant throughout the simulation period.

2.3.3 Model initialisation

The simulation runs with the model ANIMO are split into two simulation periods, an initialisation run and a scenario run. The initialisation run is necessary to calculate the accumulated amounts of phosphate in the soil due to manure additions in the past. Because phosphate is very strong retarded by the soil a realistic time period to add the historical phosphate load to the soil system is needed. If the time period is to short the annual phosphate additions will be so high that the yearly precipitation will be too small to dissolve all added phosphate. As a result a too large amount of phosphate will accumulate at the soil surface as insoluble phosphate salts. This will give an unrealistic vertical distribution pattern and in turn will give unrealistic leaching fluxes to surface waters during the scenario run. Therefore a period of 45 years was used to add the historical load of phosphate to the soil. The amount of addition was increased linear during the initialization period, starting with a surplus of 55 kg P_2O_5 ha^{-1} a^{-1}. Data on historical phosphate loads are given in paragraph 3.4. These loads are given as a surplus, calculated from historical loads and historical land use data up to the year 1990. The surplus of phosphate is equal to the total load of phosphate minus the amount of phosphate withdrawn from the soil with the harvest of crops or by grazing. Input to ANIMO is however the total phosphate load and processes as plant uptake, losses by harvest and grazing are calculated by the model. To arrive at the desired surplus, uptake by crops and losses were estimated and the values of the model parameters regulating these processes were calibrated to achieve the specified crop uptake and losses.

2.3.4 Scenario runs

Starting point for the scenario runs is the phosphate accumulation at each plot as calculated during the initialisation run. For the future scenarios the land use was taken constant for the whole simulation period and is set to present land use.

Within the Dutch manure policy target loads of phosphate are expressed in terms of fertilization surpluses. Therefore scenario runs were based on fertilization surpluses rather then loads. This means that phosphate losses by harvesting and grazing had to be estimated beforehand an that the model had to be calibrated to arrive at the desired surplus.

Two scenarios were simulated: a scenario with a surplus of 10 kg $P_2O_5.ha^{-1}.a^{-1}$ and a scenario with a surplus of 40 kg $P_2O_5.ha^{-1}.a^{-1}$. Scenario runs were carried out for a period of 60 years.

3 Input data

Phosphate leaching was assessed for units consisting of a combination of soil type, ground water regime, historical phosphate load, land use and upward seepage. These units were created by a digital overlay procedure of the maps that hold the respective information. After aggregation and simplification, a total of 2148 different units were used for the simulations.

In this chapter this geographical information will be briefly discussed. Many of the data used in this study have been used and extensively described in other studies. In such cases, the reader will be referred to the relevant publications for a detailed description of these data.

3.1 Soil related data

In this section the soil related data are discussed; a description will be provided of the geographical distribution of soils and of the chemical and physical characterization and schematization of these soils.

3.1.1 The soil map

For a regional characterization of the soils, the soil map of the Netherlands at scale 1 : 50 000 was used (De Bakker and Schelling, 1989). This soil map distinguishes 19 different soil orders which are further subdivided using soil characteristics such as peat origin, texture of the topsoil, hydromorphic properties etc. into suborders, groups and subgroups. This study was limited to the areas in the Netherlands with sandy soils and a surplus of phosphate fertilization. This area is shown on Map 1. Within this area, approximately 200 different sandy soils types have been distinguished on the soil map.

3.1.2 Chemical characterization

For the simulation of the phosphate balance in the soil with the ANIMO model, several chemical parameters must be known. The most important are: oxalate extractable Al + Fe, natural phosphate content, pH and organic matter content.

The aluminium and iron data were derived from the Dutch Soil Information System. In this system measured data are stored per soil type and layer. Average iron and aluminium contents were computed for 72 different soil groups for each of the layers distinguished in the soil profile schematization for ANIMO. A full description of

Map 1 Location of the area studied

iron and aluminium contents in Dutch soils is provided by Reijerink et al., (1993). To reduce the number of computations, soils with similar iron and aluminium contents in the various soil layers were grouped. In this way the number of soil groups with different iron and aluminium contents was reduced to 18. For each of these groups, bulk density and natural phosphate content were also derived from the Dutch Soil Information System. Because insufficient information was available for a regional differentiation of pH values for agricultural soils, a fixed pH value was used for all soils assuming sufficient fertilization and liming.

3.1.3 Physical characterization

Water retention and hydraulic conductivity characteristics for Dutch soils have been published for 18 different topsoils and 18 subsoils (Wösten et al.,1994). These are the so called physical building blocks. Each of these building blocks is characterized by its silt content, clay content, organic matter content and median value of the grain size distribution of the sand fraction. Wösten et al. (1988) related the soil units on the soil map of the Netherlands at scale 1 : 250 000 to 21 generalized soil physical units consisting of a combination of one physical building block for the topsoil and one or more physical building blocks for the subsoil. In our study the relationship between the soil units on the soil map at scale 1 : 50 000 and those on the soil map at scale 1 : 250 000 was used to assign a soil physical unit to each of the 1 : 50 000 soil units. Because assessments were made only for that part of the Netherlands that is dominated by sandy soils and because some similar soil physical soil units were grouped, 6 out of the 21 generalized soil physical units were used for the physical characterization of the sandy soils. The grouping of similar units was based on the water holding capacity of the topsoil and subsoil because these are the parameters used in the models. A description of the 6 (groups of) units is provided in Table 2.

Table 2 Description of soil physical units

Soil physical unit	Description
7	Soils developed in wind blown sand
8,9,10,12,13	Podzolic soils in fine sand, Podzolic soils in slightly loamy find sand, Podzolic soils in slightly loamy find sand on coarse sand, Umbric Podzol soils ('Enkeerd') in slightly loamy fine sand and Gleysoils ('Beekeerd') in loamy fine sand
11	Podzolic soils in loamy find sand on loam or boulder clay
14	Podzolic soils in coarse sand
19	Sandy soils with clayey topsoil
21	Loamy soils

For each of these 6 generalized soil physical units, water retention characteristics were specified for the hydrological assessments with the WATBAL model and for the computations with ANIMO.

3.2 Ground water regime

To characterize the ground water regime of the soil units of the soil map at scale 1 : 50 000, each unit has an associated ground water class (Gt). Ground water regimes have been grouped into 11 classes that are characterized by a unique combination of average highest (GHG) and average lowest ground water (GLG) levels (see Table 3). In this study several classes were grouped to arrive at 4 different ground water regime classes (Table 4). Because high ground water levels are more important with respect to phosphate leaching than low ground water levels, grouping of classes was mainly based on similar GHG levels.

Table 3 Definition of ground water regime classes

Gt	GHG cm below surface	GLG cm below surface
I	-	< 50
II	-	50- 80
IIb	25- 40	50- 80
III	< 40	80- 120
IIIb	25- 40	80- 120
IV	> 40	80- 120
V	< 40	> 120
Vb	25- 40	> 120
VI	40- 80	> 120
VII	80- 140	> 120
VIII	> 140	> 120

In Table 4 the average highest and lowest ground water levels are given that have been used to simulate the hydrology based on recent measured data. It should be noted that due to recent improvement of drainage conditions of wet soils, average highest ground water level for class 4 is outside the range provided in Table 3. An extensive discussion on this subject is provided in Reijerink et al. (1993).

Table 4 Ground water regime classes used to simulate the hydrology with WATBAL

Generalized class	Ground water classes	GHG cm below surface	GLG cm below surface	Surface area (%)
1	II	21	66	2.4
2	IIb, III, IIIb	34	103	43.0
3	IV	60	104	8.8
4	V, Vb	50	139	45.8

Because phosphate leaching from soils with a low average highest ground water table is expected to be negligible compared to soils with shallow average highest ground water tables, only the generalized classes 1 to and including 4 were used in the computations (Schoumans and Kruijne, 1995).

3.3 Land use data

Land use was assigned to soil types by overlaying the soil map of the Netherlands with the LGN land use map. This land use map was established on the basis of satellite images from LANDSAT-TM and SPOT for pixels of 25*25 m. In this way the distribution of land use over soil types was established separately for each of the sheets of the soil map. To distinguish between agricultural land use and other land use, the less detailed BARS map was used. This map shows delineations of different land use types such as nature, industry and urban areas. The procedure to assign land use types to soil types is extensively described in Reijerink et al. (1993). In our study calculations were made for three different land use types: grass, maize and other annual crops such as potato's and sugar beets.

3.4 Historicalal phosphate load

The historical phosphate load was estimated for gridcells of 2.5 * 2.5 km (Reijerink et al. 1993). For the period before 1970 an average phosphate load for the entire area was used. For the period 1970-1990 phosphate loads from animal manure were computed with a model that takes account of the number of livestock per farm aggregated for 2.5 * 2.5 km gridcells and the manure production per livestock category. Manure is distributed over the agricultural land in several fertilization rounds until all manure is distributed. In this procedure manure surpluses are mostly assigned to grass and maize. Details on the procedure and figures on manure productions can be found in Reijerink et al. (1993). The historical cumulative phosphate load was divided into 6 classes for each land use type (Table 5) in such a way that each class refers to an agricultural area comparable to the other classes. In the simulations the area weighted mean value for each class was used as the cumulative historical phosphate load.

Table 5 Classes of historical cumulative phosphate loads (kg $P_2O_5.ha^{-1}$) for the period before 1990

Class	Maize		Grass and other land use	
	boundaries	average	boundaries	average
1	< 4000	3096	< 2000	1948
2	4000- 6000	5123	2000-2500	2253
3	6000- 8000	6975	2500-3000	2761
4	8000-10000	8721	3000-3500	3214
5	10000-12000	10635	3500-4000	3708
6	> 12000	13504	> 4000	4396

3.5 Upward seepage

Upward seepage was derived from calculations with the NAGROM ground water model. This model was applied to a large number of plots in the Netherlands and yields (amongst other parameters) the upward or downward flux at a depth of 7 m below soil surface. For this study, these fluxes were divided into 5 classes (Table 6). A map was created showing the areal distribution of each of these classes. This map was overlayed with the other maps to assign a seepage class to each unit for which calculations were made with ANIMO. In the assessment of the hydrology with the WATBAL model the average seepage value per seepage class was used.

Table 6 Classes of upward seepage ($mm.d^{-1}$)

Class	Upward seepage
0	no upward seepage
1	< 0.3
2	0.3-0.6
3	0.6-0.9
4	0.9-1.2

The concentration of organic and inorganic P in seepage water was set to 5.10^{-6} kg P m^{-3} and 5.10^{-5} kg P m^{-3} respectively.

4 Model validation on the Schuitenbeek catchment

4.1 Introduction

To obtain some insight in the validity of the model calculations, the procedure to compute phosphate leaching from agricultural soils was tested on the intensively monitored Schuitenbeek catchment. The location of this catchment is given in Map 2. In this catchment, measurements have been made of (amongst other parameters) phosphate leaching and phosphate concentration for the period 1990-1993 (Schoumans and Kruijne, 1995). It therefore provides an opportunity to compare measured and simulated phosphate-related parameters which provides insight in the validity of the model results.

Map 2 Location of the Schuitenbeek catchment

4.2 Method

To compare measured and simulated data, the modelling procedure had to be adapted with respect to the phosphate fertilization after the initialization period. In the procedure for the entire area with sandy soils, the model was initialized using the period 1945-1990. Thereafter two fertilization scenario's were applied one with 10 kg phosphate surplus and one with 40 kg phosphate surplus. In comparing measured and simulated data for the year 1990-1993, however, these scenarios cannot be used because the actual phosphate surplus was higher than the surplus in the scenarios.

Therefore, a 'realistic' fertilization scenario was made for the Schuitenbeek catchment. In this scenario, phosphate fertilization was assumed to equal the allowed maximum fertilization rates defined by the Dutch manure regulations. Fertilization for 1990-1994 is shown in Table 7 (Schoumans and Kruijne, 1995). Fertilization after 1994 was assumed to equal to that in 1994.

Table 7 Yearly phosphate fertilization rates for 1990-1994 (kg $P_2O_5.ha^{-1}.a^{-1}$).

Year	Grass	Maize
1990	250	350
1991	200	250
1992	200	250
1993	200	200
1994	200	150

For the entire sandy region, the ground water model was calibrated using one area independent value for the average highest ground water levels (Table 4). For the Schuitenbeek region, however, actual information on average highest and average lowest levels is available for the ground water regimes within the region. This information shows the average highest ground water level of 50 cm for ground water level class V (representative for all ground water table classes V and Vb in the Netherlands) to be significantly deeper than the measured average highest ground water levels of 20-38 cm in the Schuitenbeek catchment. Therefore, the WATBAL model was calibrated using the average highest ground water level of 33 cm, which represents Humic/Gleyic Podzol soils ('Veldpodzol'), the dominant soil with Gt V in the catchment.

Estimated uptake of phosphate is about 100 kg $P_2O_5.ha^{-1}.a^{-1}$ for grass and 70 kg $P_2O_5.ha^{-1}.a^{-1}$ for maize resulting in higher surpluses than the surpluses of both scenarios.

Unfortunately, comparison of simulated and measured data on leaching is hampered by the fact that the water discharge used for the simulations was calculated for a 15 years period that does not contain the years in which P leaching was measured. Therefore, we did not compare leaching values from specific years. Only the average values for the measurement period 1990-1993 were compared with the moving average computed at 1998, the midpoint of the simulated period 1990-2014.

4.3 Comparison of measured and simulated phosphate saturation

To evaluate the results of the initialization period in which the historical phosphate fertilization was applied, the simulated ratio between oxalate extractable phosphate and the sum of oxalate extractable iron plus aluminium as a function of depth was compared to measured ratios (Breeuwsma et al., 1989). A comparison was made between measured data from wet grassland soils and the simulation results of grassland with a generalized ground water regime class of 1 or 2 (cf. Section 3.2). Figure 1 shows the average degree of phosphate saturation (DPS = P/(0.5(Al+Fe)) * 100%) for these wet grassland soils. Simulated phosphate saturation is within the range of measured data. Average values for the subsoil are close to measured average values. In the topsoil the saturation is somewhat overestimated in the upper 20 cm whereas in the layer between 20 and 50 cm the phosphate saturation is underestimated thus the simulated phosphate front is sharper than the measured front. This may be due to the way manure application was modelled. Manure was added to the soil surface and the soil was not ploughed. In other words the mixing effect of land management on P redistribution was not taken into account. It should be noted, that the schematization of the soil profile for ANIMO using layers of 15 to 20 cm thickness somewhat hampers the comparison of the simulated phosphate front with the measured front. Nevertheless, there is a fair agreement between computed and measured data .

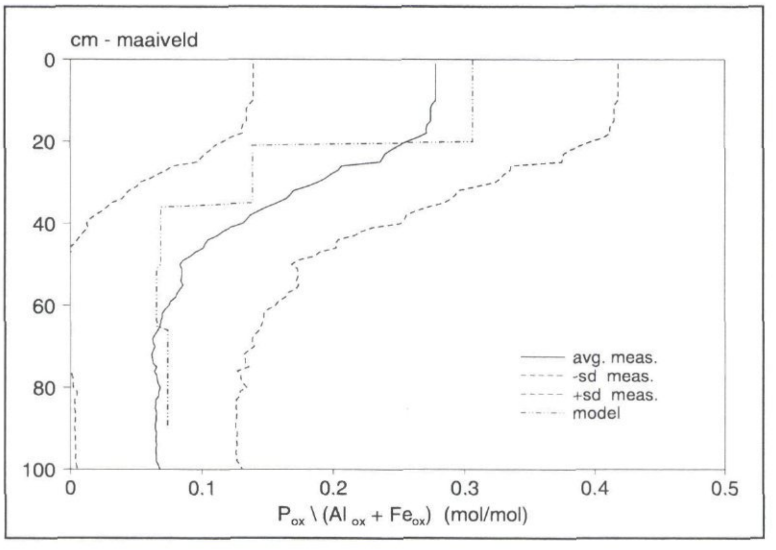

Fig. 1 Measured and simulated degree of phosphate saturaton (DPS)

4.4 Comparison of measured and simulated phosphate discharges and concentrations

Phosphate discharges and concentrations of the Schuitenbeek have been measured at the outlet of the catchment area in the period 1990-1994. Details on the results of the various measurements can be found in Schoumans and Kruijne (1995). Table 8 gives the measured discharges and concentrations between 1990 and 1993.

Table 8 Measured data for the Schuitenbeek catchment

Year	P concentration (mg $P.l^{-1}$)	P discharge (10^6 kg P)	Water discharge (10^6 m^3)
1990	0.42	2.74	6.5
1991	0.38	1.92	5.0
1992	0.41	3.05	7.4
1993	0.52	5.97	11.4

The simulated values for areas with grass and maize are provided in Table 9. In this table simulated data have been grouped for each combination of land use and generalized groundwater regime class (GWC). Average values weighted by area and total P discharge for the entire catchment are provided at the end of the table.

The calculated average water discharge for 1990-1993 was $9.4*10^6$ m^3, which is in good agreement with the average measured discharge of $10*10^6$ m^3 for the period 1977-1987 (Vermulst, 1993) despite the variations in discharge in both periods. The total water discharge from the region is thus fairly well assessed. This does not necessarily mean however, that the hydrology is correct with respect to the distribution of the discharge over the three drainage levels.

Comparison of Tables 9 and 10 show that for the wet soils (GWC 1 and 2) P concentrations are similar or somewhat higher than the measured concentration in the Schuitenbeek. For grassland with generalized ground water regime class 4 the phosphate concentration is much lower. Compared to the study of Schoumans and Kruijne (1995) who also validated the model ANIMO with model calculations for the Schuitenbeek catchment for the period 1990-1993, the discharge from grassland with Gt III and Gt V in this study is low. The reason for this difference is most likely the difference in the schematization of the ground water regime classes. In the simulations of Schoumans and Kruijne most of the grassland is assigned to a ground water regime class with a GHG (average highest ground water level) shallower than 25 cm below soil surface whereas in this study grassland was assigned mainly to GWC 2 and 4 with GHG's of 33 cm below soil surface. Furthermore Schoumans and Kruijne used a layer thickness of 25 cm for the first two layers whereas we use a layer thickness of 15 cm for the first layer and 20 cm for the second layer. The smaller layer thickness causes a lower dispersion and hence a sharper phosphate front. It was already concluded that the simulated front for this study was too sharp which causes lower leaching fluxes. For ground water table class 2 and 4 the ground water level only occasionally enters the top layer so that leaching will be low. The combination of grass and GWC 4, however, occupies about 40% of the assessed area of the catchment which causes the weighted average P concentration to be 0.29. This is about 30% lower than the measured values of about 0.4-0.5.

The computed average P discharge over the simulation period is $306*10^6$ kg. This is about 10% lower then the measured average of $3.42*10^6$ kg P.

It can be concluded that concentrations and discharge for the Schuitenbeek catchment are somewhat underestimated by the model. This is possibly due to the simulated phosphate front which is sharper than the measured front.

Table 9 Simulated phosphate discharge and concentrations in the Schuitenbeek catchment

GWC	Land use	P discharge ($kg\ P_2O_5.ha^{-1}.a^{-1}$)	P concentration ($mg\ P.l^{-1}$)	Area (ha)	Total P discharge (10^6 kg)
1	maize	9.22	0.91	7	0.03
2	maize	6.77	0.67	619	0.18
4	maize	3.66	0.32	858	0.32
1	grass	4.88	0.46	1135	0.24
2	grass	3.70	0.36	9029	1.45
4	grass	1.95	0.16	9847	0.84
Total				21497	3.06
Average		3.16	0.29		

5 Results and discussion

In this chapter the results of the scenario simulations are presented. In the first part of this chapter, the results from some particular plots are described in more detail than the results of the scenario runs for the whole region to get more insight in the processes regulating phosphate leaching to surface waters. In the second part of this chapter the results for the entire sandy region are described and discussed.

5.1 Results of some particular plots

For some plots (a plot is a unique combination of ground water class, seepage class, soil physical unit, soil chemical unit and land use) we analyzed the model output for each plot separately. This was done to obtain more detailed information of the balance terms of the processes included in the model and thus facilitate the interpretation of the clustered output as used on the regional level. Results of the plot calculations were also used to determine the length of the time period for the regional scenario analysis. The time period for the plot scenario calculations was 120 years, together with the initialisation the total simulation period was 165 years. Plot calculations were carried out for maize and grassland for the ground water level regime classes III and V, the most common classes in sandy areas. Table 10 gives an overview of the plot calculations, and gives phosphate saturation at the end of the initialisation run (start of the scenario run).

Table 10 Some characteristics of the plots

Plot	Ground water class	Land use	Soil type	Historicalal load (kg $P_2O_5.ha^{-1}$)	Degree of phosphate saturation (DPS)
P1	Gt III	maize	humic podzol	8721	81
P2	Gt III	maize	humic podzol	5123	65
P3	Gt III	maize	humic podzol	3096	51
P4	Gt V	maize	humic podzol	8721	67
P5	Gt III	grass	umbric gley soil	3214	40
P6	Gt III	grass	umbric gley soil	2253	39
P7	Gt V	grass	umbric gley soil	3214	35
P8	Gt III	grass	humic podzol	3214	42

To obtain insight in trends, running averages were used in the presentation of phosphate fluxes. This running average is the total flux averaged over 15 years. This means an average over the period starting 7 years before the year for which the value is presented and ending 7 years after that particular year. In this way fluctuations due to the hydrology are suppressed.

First the results for plot P1 are discussed. The other plots are not discussed separately, but only differences between the plots are highlighted. Appendix 2 contains the complete set of figures of the plot calculations. Figure 2 gives the leaching fluxes at the GHG level for plot P1, maize on a humic podzol (veldpodzol) with ground water regime class III and a cumulative historical load of 8721 kg $P_2O_5.ha^{-1}$, for the +10 (scenario A) and

+40 kg surplus (scenario B) scenarios. The historicalal load is quite extreme for a soil with such undeep groundwater levels. Figure 3 gives the leaching fluxes towards surface waters for both scenarios.

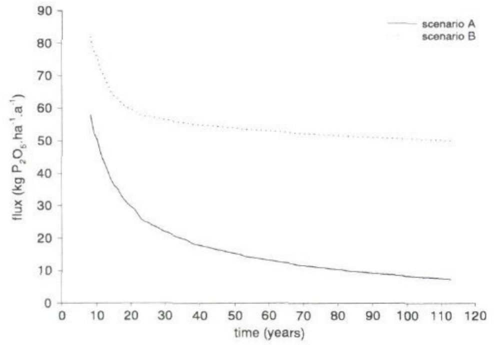

Fig. 2 Plot P1: leaching fluxes at GHG-level for the +10 kg (A) and +40 kg scenario (B)

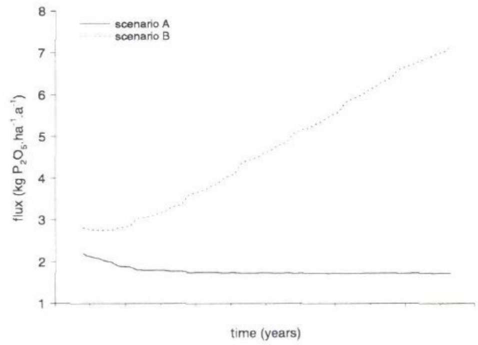

Fig. 3 Plot P1. P leaching towards surface waters for the +10 kg scenario (A) and +40 kg scenario (B)

The leaching flux at the GHG-level decreases for both scenarios to the level of the computed surplus. This decrease is caused by the fact that the soil was already very strong saturated up to GHG level and the surplus before the start of the scenario simulations was higher than the surplus of both scenarios. Figure 3 shows for the +40 kg surplus scenario a deterioration of the situation. Yearly leaching fluxes towards surface waters increase during the whole simulation period. The +10 kg scenario shows a consolidation of the present leaching fluxes. Figures 4A-4D give the various terms of the phosphate balance for the upper part of the profile above GHG level (Fig. 4A (+10 kg scenario) and Fig. 4C (+40 kg scenario) and for the profile below GHG level (4B (+10 kg scenario) and Fig 4C (+40 kg)). The different terms are:

- Input: for the profile above GHG this is the phosphate surplus (or net input) which is calculated as the remainder of the total amount of P added to the soil minus the amount withdrawn from the soil by harvesting (net P uptake by the crop). For the profile below GHG this is the downward leaching flux at GHG level;
- Fixation of mineral P is the flux of P withdrawn from the soil solution by the slow diffusion reaction which is practically irreversible;
- Sorption of mineral P represents the flux of the fast surface reaction which is reversible. A positive flux indicates sorption whereas a negative flux indicates desorption;
- Immobilization of organic P is the change of the amount of insoluble organic matter stored in the soil, a positive flux indicates immobilisation whereas a negative flux indicates net mineralization (a decrease in the stock of soil organic matter);
- The resultant of these processes is presented by the net flux:

$$P_{netflux} = P_{surplus} - (P_{fixation} + P_{sorption} + P_{immobilisation}) \tag{8}$$

For the profile above GHG level this net flux almost equals the downward flux at the GHG level (changes in soil solution storage are not accounted for in this balance). For the profile below the GHG level the net flux is almost equal to the sum of the leaching flux towards surface waters and the leaching flux at the bottom

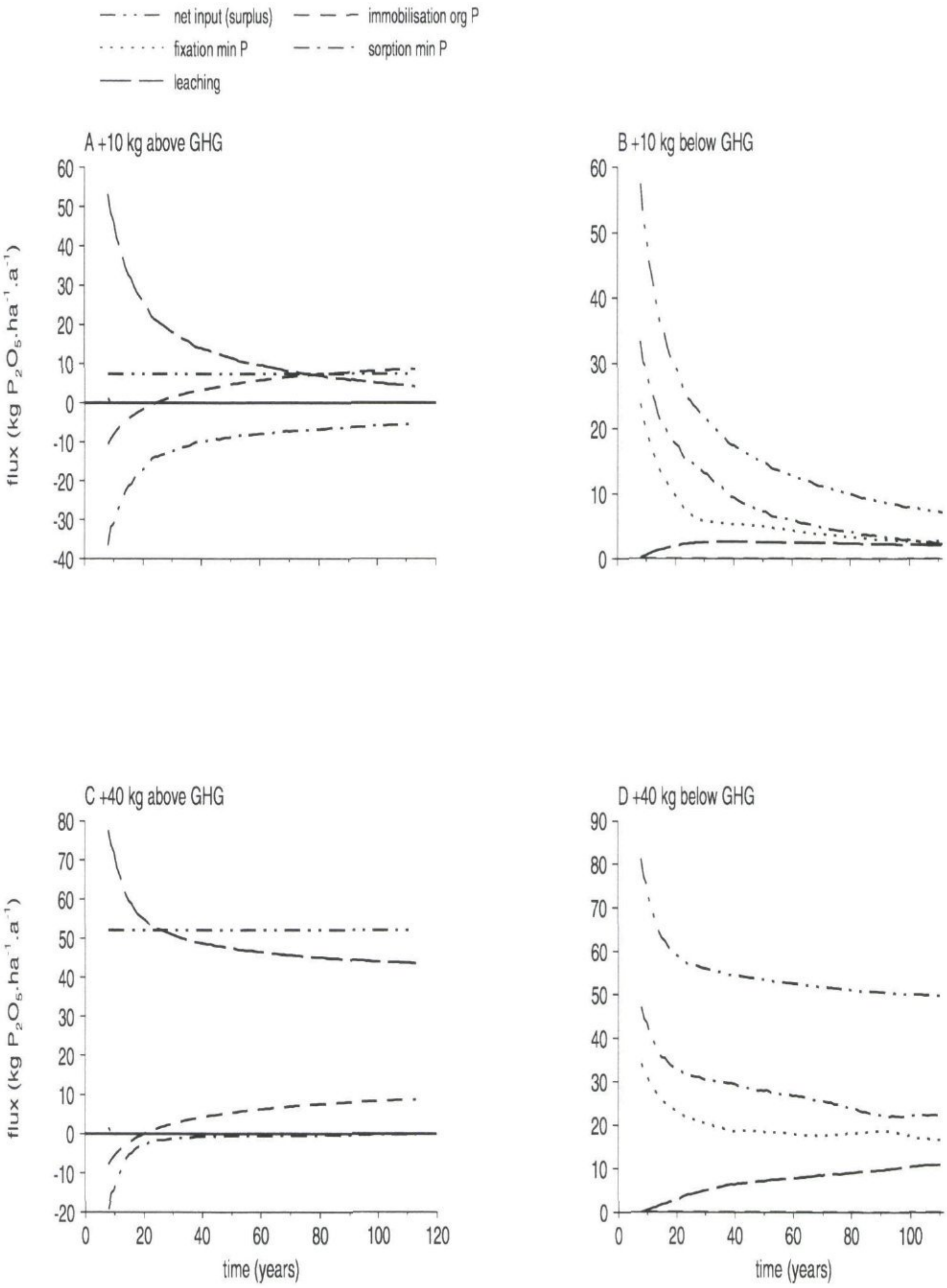

Fig. 4 Plot P1: Balance terms for the profile above GHG; A +10 kg scenario and C +40 kg scenario, balance terms for the profile below GHG; B +10 kg scenario and D + 40 kg scenario.

of the profile. Leaching at the bottom of the profile is very small if not negative (in the case of seepage). In other words : $P_{leaching} \approx P_{net\ flux}$

For both scenarios almost the entire manure surplus leaches to soil water at GHG level. For scenario A ,the +10 kg surplus scenario, leaching at GHG level is higher than the surplus due to net mineralization of organic matter during the first twenty years and desorption of inorganic P. For scenario B, the + 40 kg scenario , there is first also a period of net mineralization and desorption but after 10 years these terms are relatively low compared to the surplus. At the end of the simulation period a part of the surplus is retained in the soil above GHG level for both scenarios as a result of net immobilization of organic matter. Below the GHG level most of the surplus is immobilized by adsorption and fixation for scenario A, the net flux is even a little negative, whereas for scenario B there is first a period in which the complete surplus of phosphate is immobilized but after some time part of the surplus phosphate is not immobilized anymore and leaching towards surface waters or deeper soil water increases.

Effect of historicalal load on leaching fluxes.

Figures 5 and 6 give the leaching fluxes at GHG level for the plots P1, P2 and P3 (see Table 10 for properties of the different plots), which are the same with respect to soil type, crop (maize) and hydrology but differ in historicalal load.

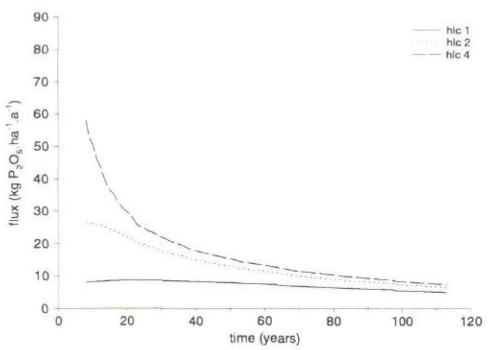

Fig. 5 Effect of historical load on leaching fluxes at GHG level +10 kg scenario

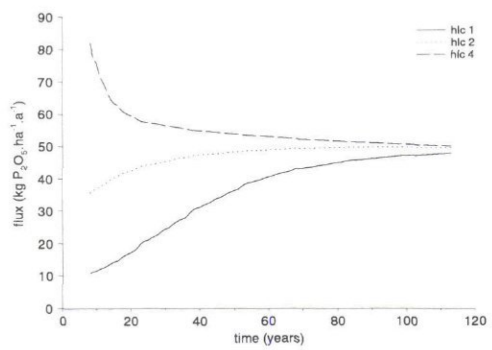

Fig. 6 Effect of historical load on leaching fluxes at GHG level for the +40 kg scenario

The leaching flux at GHG level for all three plots with the +10 kg scenario decreases towards the level of the surplus. The decrease is a result of desorption and net mineralization of organic P. For the +40 kg scenario leaching fluxes increase for the historicalal load classes 1 and 2 and decrease for historicalal load class hlc4. This difference is caused by the difference in phosphate saturation. For the two plots with the lower historicalal loads part of the surplus added to the soil is withdrawn by sorption and fixation.

Leaching towards surface waters decreases initially for all of the three historicalal load classes for the + 10 kg scenario and after some time (20 years) leaching fluxes remain constant. By that time most of the input to the soil profile below GHG level (this is the downward leaching flux at GHG level) is whitdrawn by sorption and fixation. The levels of leaching towards surface waters remains different for the three historicalal load classes. In case of the +40 kg scenario leaching fluxes increase for all three historicalal load classes.

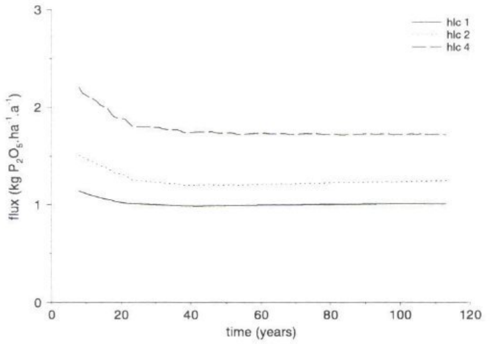

Fig. 7 Leaching towards surface waters for different historical loads +10 kg scenario

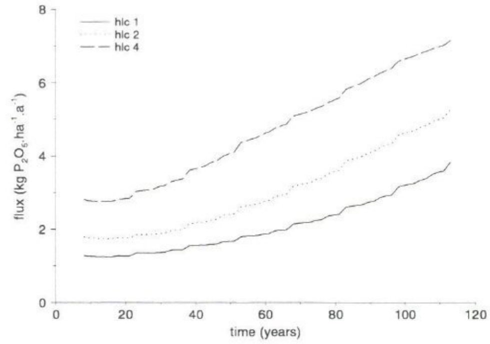

Fig. 8 Leaching towards surface waters for different historical loads +40 kg scenario

For grass differences in leaching fluxes are very small due to the relatively small differences in historicalal loads for the historicalal load classes.

Effect of GHG level on leaching fluxes

Figures 9 and 10 show the effect of the level of the average highest groundwater level on the leaching fluxes at GHG level by comparing the fluxes of plot P1 with plot P4. Plot P1 and plot P4 only differ in GHG, other characteristics were the same. The P front for the plot with the lower GHG is more retarded due to a higher phosphate sorption capacity of the profile above GHG level. The most striking difference between the two groundwater classes is the initial increase of P leaching to a level higher than the added surplus followed by a decrease towards the surplus level (figure 9). The increase is caused by desorption in the toplayers of the profile.

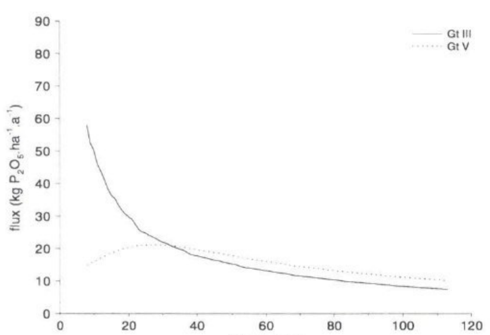

Fig. 9 Leaching fluxes at GHG level for different Gt's +10 kg scenario

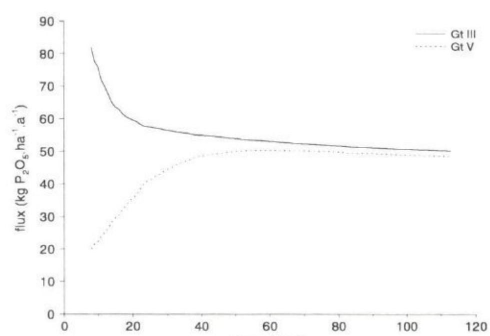

Fig. 10 Leaching at GHG level for different Gt's +40 kg scenario

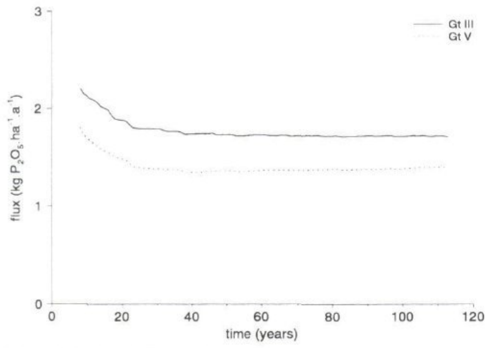

Fig. 11 Leaching fluxes towards surface waters for different Gt's, +10 kg scenario

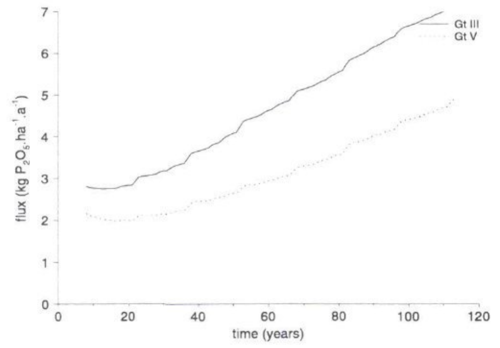

Fig. 12 Leaching fluxes towards surface waters for different Gt's, +40 kg scenario

Leaching fluxes towards surface waters are lower for the plot with the lower Gt (Gt V) for both scenarios due to the lower groundwater levels of this plot.

Effect of different crops

Most important differences between maize and grass are the deeper rootzone for maize which is 40 cm while for grass the rootzone is only 20 cm. In ANIMO organic matter production by grass is added to the soil only by root decay and harvest losses, for maize also root exudates are added as organic matter to the soil. There is also a difference in the formulation of root uptake by the different crops. In the case of grass the P status of the soil (P in solution and P adsorbed) determines the maximum crop uptake. For maize maximum uptake is not depending on the P status of the soil. This difference in uptake behaviour can be seen in the balance figures (see Appendix 2). The surplus for maize is constant in time whereas the surplus for plots with grassland changes in time as a result of changing crop uptake by grass.

Another striking difference is the change in soil organic matter for both crops. The plots with maize immobilize organic matter, the stock of soil organic matter increases, whereas for plots with grass the stock in soil organic matter decreases. This difference must be due to the deeper rootzone for maize and the fact that the soil is not ploughed. Death root material (and in the case of maize also root exudates) is added to the soil in the rootzone. The top twenty centimetres are better aerated than the soil between 20 and 40 cm below soil surface and thus mineralization rates will be higher in the top soil. In the case of grass all organic matter is added to these top twenty centimetres, in the case of maize part of the organic matter is also added to the second twenty centimetres. Roots for grass and maize had the same mineralization rates. The continuing immobilization of organic matter however seems not to be very realistic. This may have consequences for the results of the + 10 kg scenario for which leaching fluxes to surfaces waters remain on a constant level. In case mineralization fluxes are underestimated leaching of P at GHG level will be higher and possibly leaching fluxes towards surface waters also increase if this extra input is not immobilized by sorption and fixation.

Effect of soil type

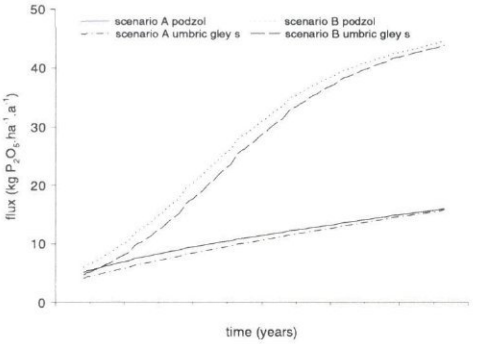

Fig. 13 Leaching fluxes at GHG level for a podzol and an umbric gley soil

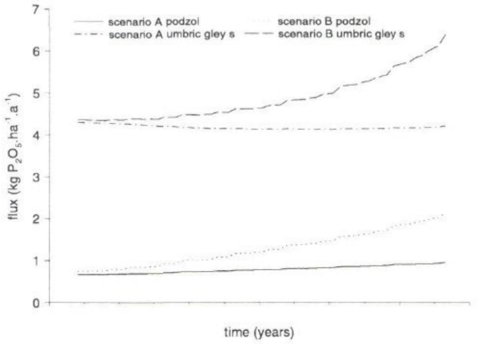

Fig. 14 Leaching towards surface waters for a podzol and an umbric gley soil

The effect of a different soil type on leaching fluxes was examined by the comparison of the results for plot P5, an umbric gley soil, and plot P8 a podzol. All other properties were the same for both plots. Figure 13 shows that leaching fluxes at GHG level are almost the same for the two different soils for both scenarios. Leaching fluxes for the podzol are a little higher than for the umbric gley soil due to a little higher phosphate saturation of the podzol. This higher phosphate saturation is the result of a smaller Al and Fe content in the top soil. Leaching fluxes towards surface waters are much higher for the umbric gley soil. Below GHG the Al and Fe content of the umbric gley soil is much lower than that of the podzol. Due to this lower Al and Fe content less phosphate is sorbed and fixated and more P is leached towards surface waters.

5.2 Results of the assessments for the entire sandy region

In this section results of the calculations made for the region with sandy soils and a manure surplus in the Netherlands will be presented and discussed. Results will be presented for the 2 scenario's described in Section 2.3.3: the scenario with a surplus of 10 kg P_2O_5.ha.$^{-1}$.a.$^{-1}$ and the scenario with a surplus of 40 kg P_2O_5.ha.$^{-1}$.a.$^{-1}$ and for two moments in time: after 20 and after 50 years of scenario simulation. The results presented are the moving average values over a time period of 15 years: the value for year 20 thus represents the average value for the period from year 13 to year 27 whereas the results for the year 50 represent the average value for the period of year 43 to year 57. Apart from the moving average, the moving 95 percentile is presented for some parameters. The 95 percentile represents the value that is exceeded in 5% of the area of a catchment and thus gives an indication of the highest phosphate loads and concentrations in a catchment.

5.2.1 Phosphate saturation

The degree of phosphate saturation of a soil was derived from the ratio between oxalate extractable P and the sum of Al and Fe in accordance with Reijerink et al., (1993):

$$DPS = 2 * \frac{P_{ox}}{Al+Fe} \tag{9}$$

Table 10 shows the computed degree of phosphate saturation as a function of land use and generalized ground water table class.

Table 11 Average degree of phosphate saturation

Land use	GWC	DPS			
		+ 10 kg		+40 kg	
		20 years	50 years	20 years	50 years
maize	1	0.78	0.78	0.79	0.80
maize	2	0.68	0.68	0.70	0.72
maize	3	0.52	0.54	0.54	0.58
maize	4	0.52	0.55	0.55	0.57
grass	1	0.47	0.46	0.51	0.53
grass	2	0.43	0.44	0.47	0.52
grass	3	0.32	0.35	0.34	0.41
grass	4	0.39	0.41	0.41	0.48

Table 10 shows that the average degree of phosphate saturation exceeds the criterium for a phosphate saturated soil: DPS > 0.25. The total percentage of phosphate saturated soils of the assessed combinations at the start of the simulations was 94%. This is close to the 89% derived from the data by Reijerink and Breeuwsma (1993). The area with strongly phosphate saturated soils (DPS > 0.5) was computed at 28%. This is somewhat lower than the 34% by Reijerink and Breeuwsma (1993); for very strongly phosphate saturated soils (DPS > 0.75) the computed areas where 6% and 13% respectively. The substantial difference in very strongly phosphate saturated soils is explained by the fact that in this study classes of historicalal phosphate load were used whereas Reijerink and Breeuwsma (1993) used the computed phosphate load separately for each gridcell. Especially the class with the highest phosphate loads, contains a very wide range in loads. Because we used the average value of each class, the load is underestimated on areas with the highest loads is underestimated. These areas, however, form the major part of the area with very strongly phosphate saturated soils.

Table 10 shows that with the +10 kg scenario, the phosphate saturation hardly increases, a small increase was found only for ground water regime classes 3 and 4. For the +40 kg scenario, an increase in phosphate saturation is observed for all combinations. This increase is higher than for the +10 kg scenario.

5.2.2 Phosphate leaching to surface waters

Phosphate leaching to surface waters was computed as the sum of the leaching of organic and mineral phosphate to the three drainage levels (cf chapter 2). To assist in the interpretation of the maps with results that will be discussed, Map 3 shows the median phosphate saturation per gridcell in 1990 computed by Reijerink and Breeuwsma (1993).

Figures 15 and 16 show the weighted average phosphate leaching as a function of time and of phosphate saturation class for the sandy area. These figures show that for the +10 kg scenario hardly any change is observed in the average phosphate leaching, whereas leaching at the +40 kg scenario shows a clear increase in time. For the +10 kg scenario almost the entire surplus is bound to the soil matrix and immobilized in organic matter (cf Section 5.1). Figures 15 and 16 also show that there is a marked increase in leaching with time if the phosphate saturation is higher than 75%.

Results of the computations as a function of land use and ground water table class are provided in Table 11. Similar to the trends in Figures 15 and 16, results in Table 11 show that for the +10 kg scenario a very small increase in leaching was calculated between year 20 and year 50. For some of the grassland soils a decrease was computed. For these units, however, the simulated surplus was less than 10 kg (between 5 kg and 7 kg). For the +40 kg scenario all combinations of land use and GWC show an increased phosphate leaching in time. For most combinations the leaching after 50 years is substantially higher than for the +10 kg scenario.
to year 27 whereas the results for the year 50 represent the average value for the period of year 43 to year 57. Apart from the moving average, the moving 95 percentile is presented for some parameters. The 95 percentile represents the value that is exceeded in 5% of the area of a catchment and thus gives an indication of the highest phosphate loads and concentrations in a catchment.

The weighted average phosphate leaching for each of the catchment areas for both scenarios and points in time are presented in Maps 4 to 7. These maps show the differences between the leaching in the various catchments. Comparing the maps with Map 3 clearly shows that highest leaching is associated with the areas where phosphate saturation is high (e.g. the 'Gelderse vallei' area and the 'Peel' area). Furthermore it should be noted that in the central sandy area leaching is relatively high compared to the other two regions because of the larger area with shallow ground water tables (especially GWC 2).

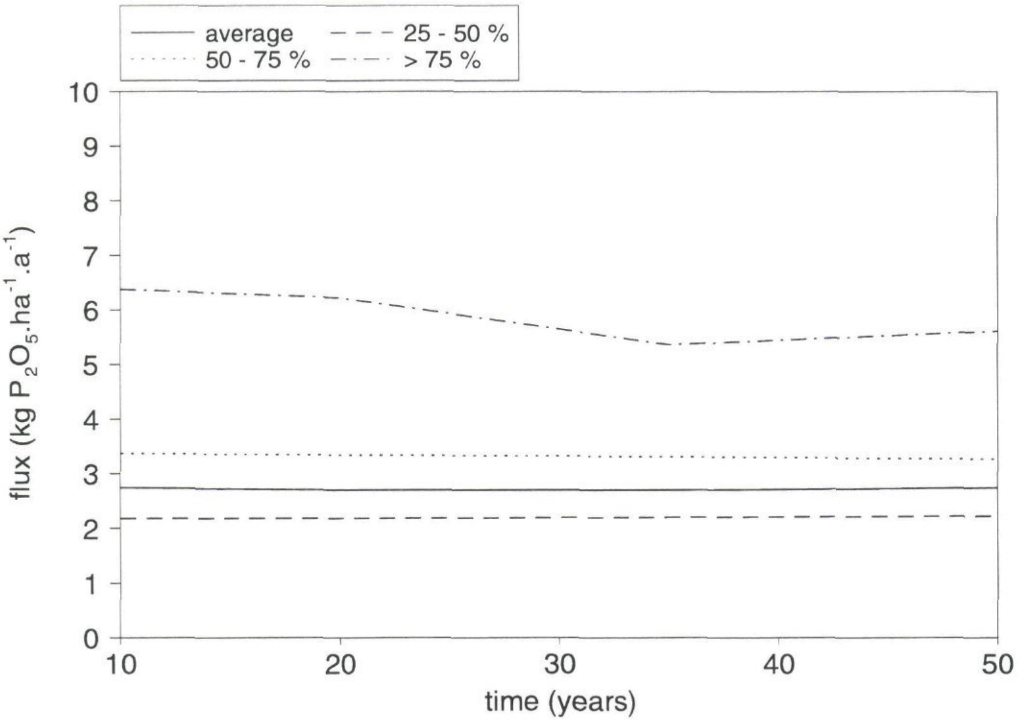

Fig. 15 Trends in average P leaching in kg P_2O_5.ha^{-1}.a^{-1} as a function of P saturation classes for the + 10 kg scenario

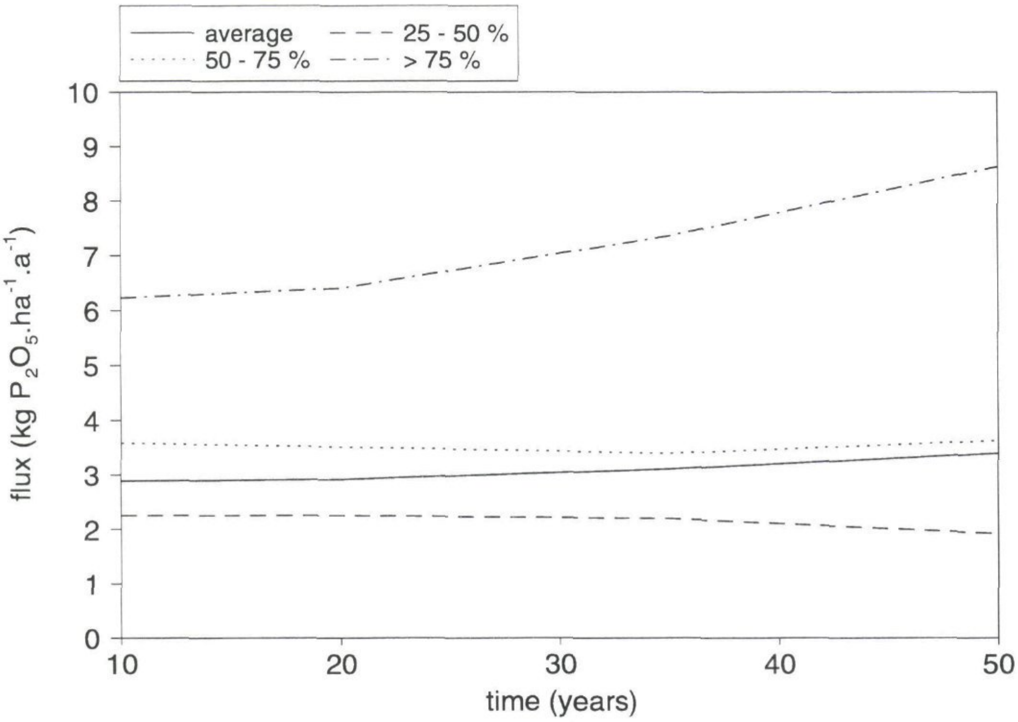

Fig. 16 Trends in average P leaching in kg P_2O_5.ha^{-1}.a^{-1} as a function of P saturation classes for the +40 kg scenario

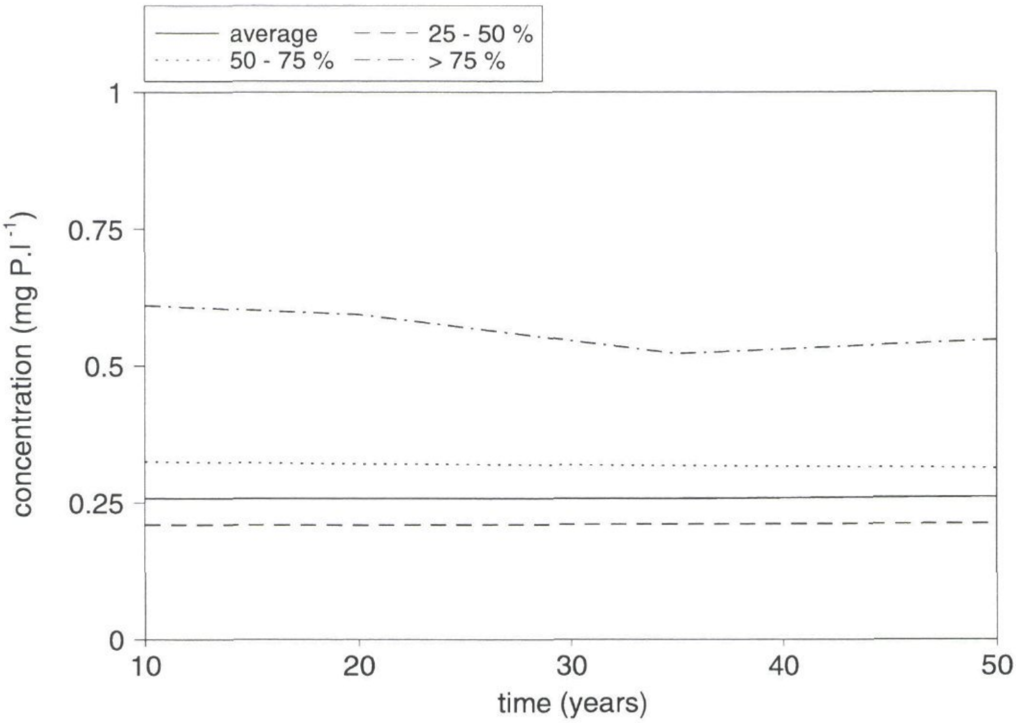

Fig. 17 Trends in P concentration (mg.l^{-1}) as a function of P saturation for the +10 kg scenario

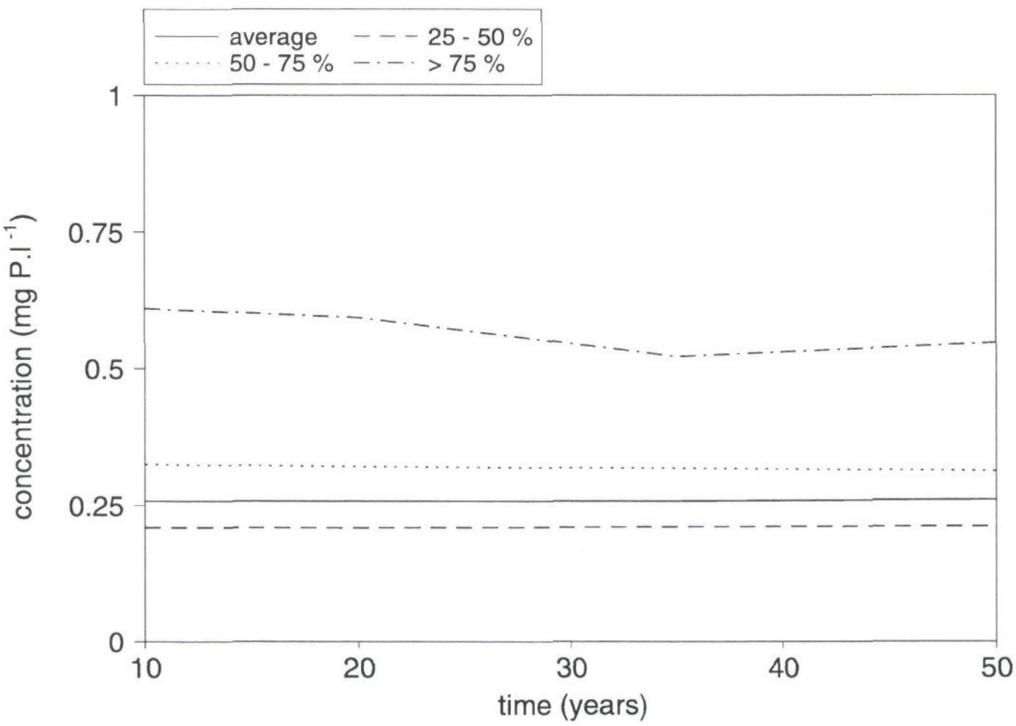

Fig. 18 Trends in P concentration (mg.l^{-1}) as a function of P saturation for the +40 kg scenario

Table 12 Average phosphate leaching (kg $P_2O_5.ha^{-1}.a^{-1}$)

Land use	GWC	Leaching			
		+ 10 kg		+40 kg	
		20 years	50 years	20 years	50 years
maize	1	7.20	7.11	9.12	11.39
maize	2	5.17	5.31	5.98	7.66
maize	3	3.39	3.57	3.72	5.04
maize	4	2.18	2.57	2.71	4.02
grass	1	3.63	3.70	3.80	4.34
grass	2	3.71	3.64	3.82	4.02
grass	3	2.73	2.66	2.75	2.71
grass	4	0.98	1.02	1.05	1.24

Because leaching occurs if the ground water level enters the phosphate saturated layer, the ground water level class has a strong influence on the leaching (see also Table 12).

It should be realized, however, that in the central sandy area ground water table class 6 occupies a large area. Leaching for this GWC was not assessed so the maps only indicate the leaching in the wetter parts of the catchment. Therefore the maps should be interpreted with care.

5.2.3 Phosphate concentration

The trend in phosphate concentrations in the leachate is presented in Figures 17 and 18. Because concentration and leaching are linearly related through the drainage flux (which does not change in the course of time) Figures 17 and 18 show similar results as Figures 15 and 16. For the +10 kg scenario the concentration hardly increases, whereas for the +40 kg scenario the average concentration increases from 0.24 $mg.l^{-1}$ to 0.31 $mg.l^{-1}$. Again very high concentrations are associated with the very strongly P saturated soils (> 75% saturation).

Weighted average phosphate concentrations in the drainage water are given in Table 13. Similar to Figure 15, for the +10 kg scenario the concentration shows hardly any increase, whereas the +40 kg scenario leads to increased concentrations in time. It should be noted that the phosphate concentration exceeds the Dutch quality standard for surface water of 0.15 $mg.l^{-1}$ for all ground water regime classes except for GWC 4.

Table 13 Average phosphate concentration in mg.l⁻¹

Land use	GWC	DPS			
		+ 10 kg		+40 kg	
		20 years	50 years	20 years	50 years
maize	1	0.61	0.60	0.76	0.94
maize	2	0.50	0.52	0.57	0.74
maize	3	0.33	0.35	0.37	0.50
maize	4	0.20	0.23	0.24	0.36
grass	1	0.33	0.33	0.34	0.39
grass	2	0.35	0.35	0.36	0.38
grass	3	0.25	0.24	0.25	0.25
grass	4	0.09	0.09	0.10	0.11

The weighted average phosphate concentrations are presented in Maps 8 to 11. These maps show essentially the same geographical patterns as maps 7 to 10. In maps 12 to 15 the 95 percentile of the phosphate concentrations are presented. These maps show the concentrations exceeded in 5% of the catchment. Maps 12 and 13 show that for the +10 kg scenario the 95 percentile of computed P concentration does not change much in time. For the +40 kg scenario (Maps 14 and 15), however, the number of catchments where the 95 percentile concentration exceeds the threshold value by a factor six or more, strongly increases.

5.3 Assessment of the effect of remedial measures to reduce phosphate leaching from phosphate saturated soils

In the previous sections of this chapter, it was shown that the positive effects of a decrease in phosphate fertilization on phosphate leaching are limited. For the scenario in which a surplus of only 10 kg P_2O_5 is allowed, leaching of P decreases for the most strongly saturated soils but the P concentrations in the leaching water remains far above the Dutch quality standard for surface waters (cf. Section 5.2.2 and 5.2.3). A more substantial decrease in phosphate discharge can only be achieved by more specific measures that reduce the phosphate leaching. Schoumans and Kruijne, (1995b) have described the effects (measured in the laboratory and in the field) of three measures to reduce phosphate leaching from agricultural soils. The most effective measures were a chemical treatment where a suspension of Fe-hydroxides was added to the topsoil (so that the additionally supplied Fe-hydroxide binds the phosphate) and a hydrological measure that changes the water flow in the soil profile in such a way that the phosphate binding capacity of the subsoil can be used. Such measures may be used to reduce leaching in areas with strongly saturated soils where a good quality of the surface water is important (such as in areas with a high nature value).

For a first tentative assessment of the effect of a chemical treatment on a regional scale, the results of the scenario assessments were used. For each of the catchments the reduction in phosphate leaching was computed assuming that the leaching of all soils with a phosphate saturation of more than 75% is reduced by approximately 60%-80% (Schoumans and Kruijne, 1995b) The effect of the chemical measure was assessed for both scenario's after 50 years of simulation. The computed moving average leaching from strongly P saturated soils for this year was reduced by 70% for the +10 kg scenario and by 78% for the +40 kg scenario and compared to the moving average of the leaching in year 8 for the +10 kg scenario which served as the 'initial situation'. For the +40 kg scenario reduction in P leaching increased because the effectiveness of the measure increases with increasing P saturation. The effectiveness was chosen in such a way that the leaching from strongly saturated soils in year 50 after the chemical treatment was the same for both scenario's.

Results for the +10 kg scenario with chemical treatment are shown on map 16. A comparison of the results from year 8 with the results of year 50 shows clearly the effect of the strong reduction in leaching from strongly P saturated soils (leaching from other soils does not change much for the +10 kg scenario). The area with strongly phosphate saturated soils is shown on map 17. A comparison of these maps clearly shows that the highest reductions of about 20% are achieved in areas with a high percentage of P saturated soils. The reduction percentage decreases from about 70% at a plot scale to 20% at a catchment scale due to the presence of less saturated soils in the latter case. For the total area a reduction of 90 000 kg was computed.

Map 18 shows the reduction in P leaching for the +40 kg scenario with a chemical treatment compared to the leaching of year 8 for the +10 kg scenario. Comparing this map to map 15 clearly shows that for the +40 kg scenario the increase in leaching for the soils with a P saturation of less than 75% diminishes the positive effects of the reduced leaching from soils with a P saturation of more than 75%. Only in a few catchments with a very high percentage of very strongly phosphate saturated soils, a decrease of leaching was computed. This shows that if specific measures for very strongly P saturated soils are taken to protect an area, e.g. with a high nature value, this treatment is only effective at the long term if the allowed P surplus for the other soils in the same area is (strongly) limited.

Results of this tentative assessments should be interpreted with care. As already pointed out in Section 5.2.1 the area with very strongly phosphate saturate soils is probably underestimated so that we may also underestimate the effect of the measures to reduce leaching on the scale of a catchment. Historical P loads were used that were computed for 2.5 *2.5 km cells. Within these cells however, fertilization may not be applied homogeneously so that P saturation in the field can deviate from the simulated one. More accurate assessments can be made if more accurate data are available on P saturation within the various areas. Furthermore, the chemical treatment has not been tested yet on an area with very different soil types and ground water regimes that may influence its effectiveness.

6 Conclusions and recommendations

Calibration of the hydrological model WATBAL did not always lead to a unique parameter set of calibrated drainage levels and drainage resistances. Computations with ANIMO however showed little difference between ANIMO runs with the different WATBAL files based on different parameter sets. Fits of simulated groundwater levels with the measured frequency distribution of groundwater levels were good for all ground water classes.

The computations were validated on the Schuitenbeek catchment by comparing simulated and measured P loads to surface waters. The comparison was somewhat hampered by the fact that meteorological data used differed from the actual weatherdata in the measurement years. Results indicate that both phosphate discharge and phosphate concentrations where underestimated. This may be caused by the fact that the simulated P front was sharper than the front measured in the field. Furthermore discharge from soils with the highest phosphate loads is expected to be underestimated due to the fact that we used only the average value of a very broad range in phosphate loads in the highest historical load class.

Plot calculations show a significant difference in the results for different historical phosphate load classes for maize. This in contrast to grassland for which differences in leaching fluxes of different historical load classes are smaller due to the smaller differences in the load of the different historical load classes. Calculations can be improved without an increase in the calculations to be done by using a higher degree of differentiation in the higher historical load classes and a lower differentiation in the lower historical load classes.

For each of the groundwater regime classes, average highest and lowest groundwater level were assumed to be independent of location. In reality however, this is not the case. More accurate assessments can be made if more information on spatial variability of the groundwater regime (e.g. the groundwater level in the course of the year) would be regionally available. This is important because the leaching of phosphate is strongly influenced by the occurrence of periods with high to very high groundwater levels in which the groundwater table enters the phosphate saturated topsoil.

In all simulations with grassland net mineralization of organic P was calculated, for maize mineralization of P occurred only in the time period at the start of the scenario calculations, hereafter organic P was immobilized. This immobilization of organic matter as calculated for the plots with maize is however questionable. Using the ploughing option of the model is expected to give more plausible results.

Plot calculations indicate that downward leaching fluxes at GHG level eventually tend to reach the level of the added surplus. At the start of the scenario simulations fluxes at GHG level can be higher than the added surplus due to net mineralization and desorption or lower due to fixation and adsorption of phosphate, depending on the degree of phosphate saturation. The plot calculations show that for scenario A (+10 kg) leaching fluxes towards surface waters are stabilized whereas for scenario B (+40 kg) leaching fluxes towards surface waters increase. In the case of the +10 kg scenario almost the entire surplus is immobilized by organic matter immobilization above GHG (only for maize) and fixation and sorption below GHG. With the + 10 kg scenario the amount of phosphate leached to surface waters is determined by the degree of phosphate saturation of the soil. The higher the degree of phosphate saturation the higher are the leaching fluxes towards surface waters. The difference in leaching fluxes between soils with different degrees of saturation does not change during the entire simulation period of 120 years. Because for the + 10 kg scenario losses are only stabilized and not substantially decreased. The +10 kg scenario can not be used to reduce phosphate leaching from (strongly) saturated soils to acceptable levels.

Different soil types with the same phosphate load may have very different levels of leaching towards surface waters due to differences in contents of Al and Fe (below GHG level).

Computations for the entire area show that phosphate concentrations and phosphate leaching hardly increase in time for the +10 kg scenario. For the +40 kg scenario however, leaching to surface waters increases significantly. For both scenarios the average phosphate concentrations exceed the Dutch quality standard for surface waters of 0.15 $mg.l^{-1}$ by a factor 1-6.

There is a marked difference in phosphate leaching and concentrations between the different phosphate saturation classes. Soils with a very high phosphate saturation (> 75%) particulary show high phosphate leaching and concentrations of almost a factor 2 higher than for soils with a degree of P saturation between 50 and 75%.

A tentative assessment of the effect of a chemical treatment to reduce phosphate leaching showed that for very strongly phosphate saturated soils such a treatment decreases the phosphate leaching in a large number of catchments. Because the treatment is applied on a relatively small area (soils with P_{sat} > 75%) effects are limited. A reduction of P leaching with more than 20% only occurs in catchments with a large area of strongly P saturated soils. However, such a reduction can only be achieved if the phosphate surplus on the other soils is low. Otherwise the increased phosphate leaching in time from these soils will diminish the positive effects of the chemical measure.

For more accurate assessments of the effect of specific measures aiming at a reduction of phosphate leaching, more information is needed on the effectiveness of these measures as a function of soil type, ground water regime and phosphate saturation. Furthermore, a more accurate assessment of the location and leaching from very strongly phosphate saturated soils is needed. This can be achieved by a similar procedure as described in this report, but more emphasis should then be put on the data collection and P related processes in the soil for such strongly saturated soils.

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Annex 1 Parametrisation of phosphate sorption

Equation 10 gives the Langmuir adsorption equation which describes the fast reversible surface reaction.

$$Q_f = \frac{K\ c\ Q_m}{1 + K\ c} \qquad (10)$$

Q_f	= amount of sorbed phosphate	($kg.m_g^{-3}$ P)
c	= phosphate concentration	($kg.m_w^{-3}$ P)
K	= Langmuir adsorption coefficient	($m_w^3.kg^{-1}$ P)
Q_m	= Langmuir adsorption maximum	($kg.m_g^{-3}$ P)

In the annotation of the units the subscripts *g* and *w* denote volumes soil and water respectively.

Q_m is calculated as;

$$Q_m = 5{,}167 \cdot 10^{-6}\ [Al{+}Fe] \qquad (11)$$

ρ	= dry bulk density	($kg.m_g^{-3}$)
[Al+Fe]	= oxalate-extractable Al and Fe	($mmol.kg^{-1}$)

Filling out equation (10) with the other parameter values gives;

$$Q_f = \frac{1129{,}0\ c\ 5{,}167 \cdot 10^{-6}\ \rho\ [Al{+}Fe]}{1 + 1129{,}0\ c} \qquad (12)$$

The slow irreversible precipitation reaction is describes as a summation of three time dependent components;

$$\frac{\delta Q_s}{\delta t} = \sum_{i=1}^{3} \alpha_i\ (K_{F,i}\ c^{N_i} - Q_{s,i}) \qquad (13)$$

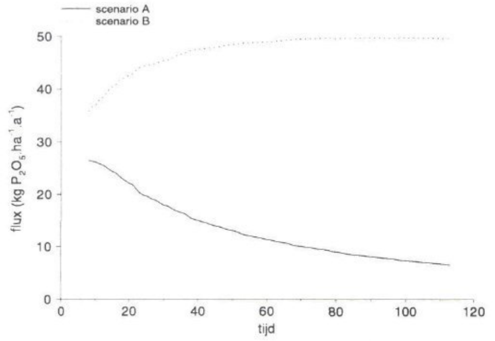
Q_s	= total amount of diffused phosphate	(kg P m_s^{-3})
$Q_{s,i}$	= amount of diffused phosphate fraction i	(kg P m_s^{-3})
c	= phosphate concentration	(kg P m_w^{-3})
N_i	= Freundlich sorption-exponent fraction i	(-)
α_i	= rate constant fraction i	(d^{-1})
$K_{F,i}$	= Freundlich sorption constant fraction i	($kg.m_g^{-3}$ P ($kg.m_w^{-3}$ P)$^{-N}$)

with the following parameter values;

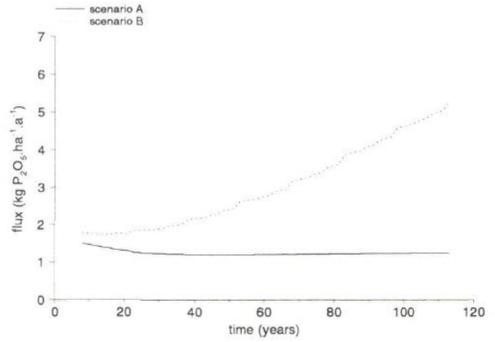
α_1	= 1,1755	(d^{-1})
N_1	= 0,5357	(-)

K_{F1} $= 11{,}87.10^{-6}\ \rho\ [Al+Fe]$ $(kg.m_g^{-3}\ P.\ (kg.m_w^{-3}\ P)^{-0{,}5357})$
α_2 $= 0{,}03340$ (d^{-1})
N_2 $= 0{,}1995$ $(-)$
K_{F2} $= 4{,}667.10^{-6}\ \rho\ [Al+Fe]$ $(kg.m_g^{-3}\ P.\ (kg.m_w^{-3}\ P)^{-0{,}1995})$
α_3 $= 0{,}0014382$ (d^{-1})
N_3 $= 0{,}2604$ $(-)$
K_{F3} $= 9{,}711.10^{-6}\ \rho\ [Al+Fe]$ $(kg.m_g^{-3}\ P.\ (kg.m_w^{-3}\ P)^{-0{,}2604})$

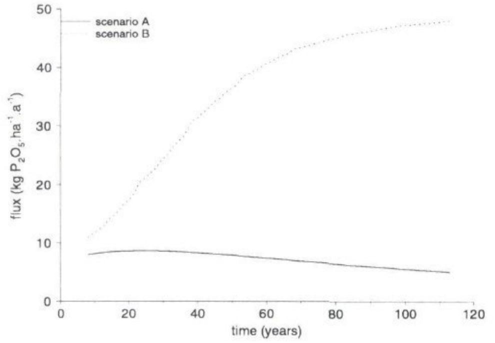
Annex 2 Results plot calculations



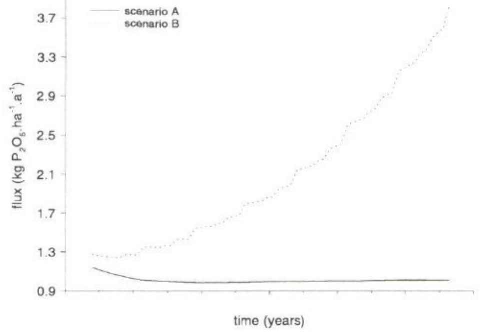
Plot P2: leaching fluxes at GHG-level for the +10 kg (A) and +40 kg scenario (B)



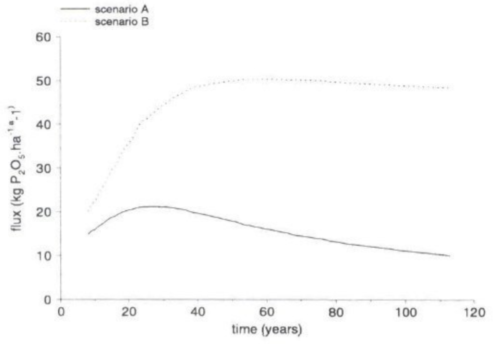
Plot P2: P leaching towards surface waters for the +10 kg (A) and +40 kg scenario (B)



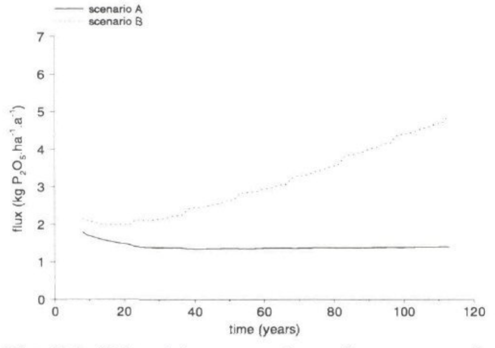
Plot P3: leaching fluxes at GHG-level for the +10 kg (A) and +40 kg scenario (B)



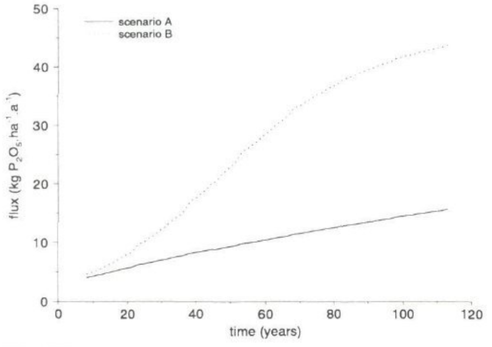
Plot P3: P leaching towards surface waters for the +10 kg (A) and +40 kg scenario (B)



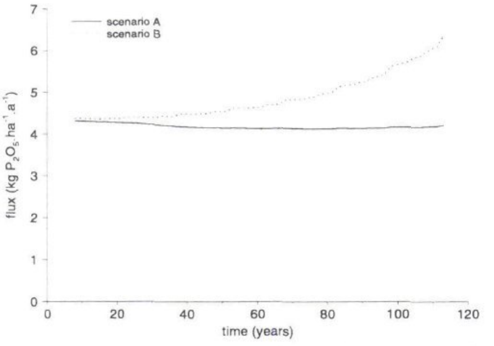
Plot P4: leaching fluxes at GHG-level fort the +10 kg (A) and +40 kg scenario (B)



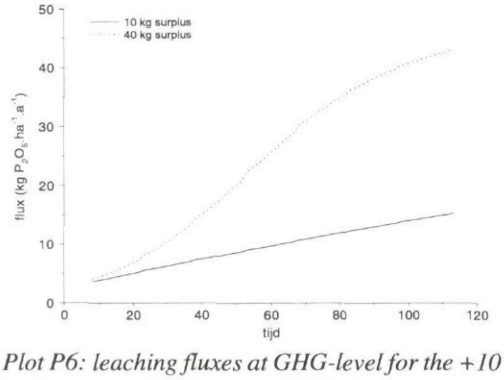
Plot P4: P leaching towards surface waters for the +10 kg (A) and +40 kg scenario (B)



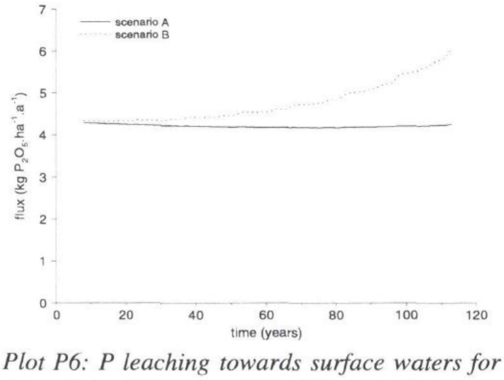
Plot P5: leaching fluxes at GHG-level for the +10 kg (A) and +40 kg scenario (B)



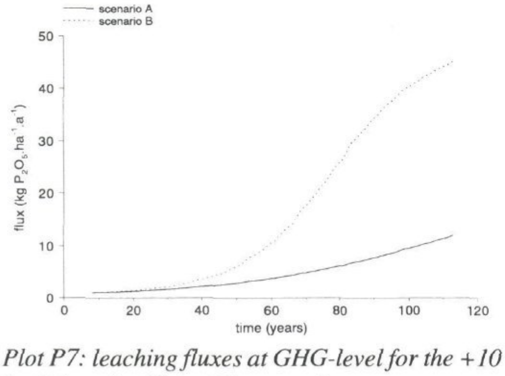
Plot P5: P leaching towards surface waters for the +10 kg (A) and +40 kg scenario (B)



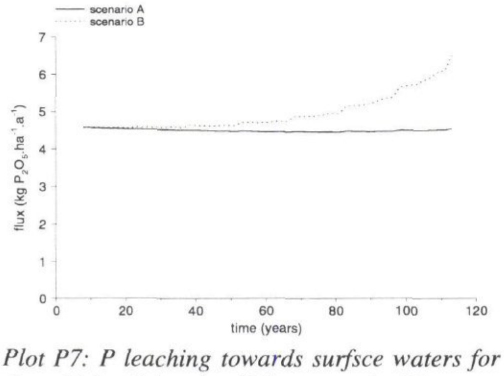
Plot P6: leaching fluxes at GHG-level for the +10 kg (A) and +40 kg scenario (B)



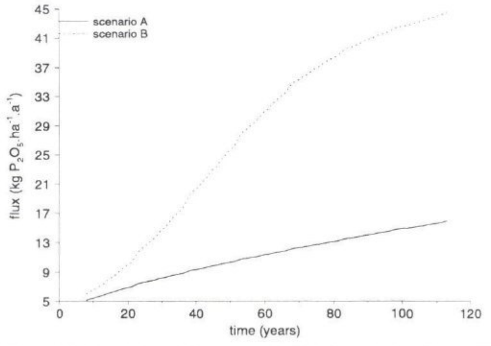
Plot P6: P leaching towards surface waters for the +10 kg (A) and +40 kg (B) scenario



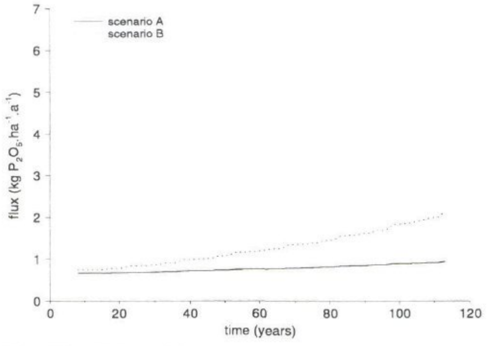
Plot P7: leaching fluxes at GHG-level for the +10 kg (A) and +40 kg (B) scenario



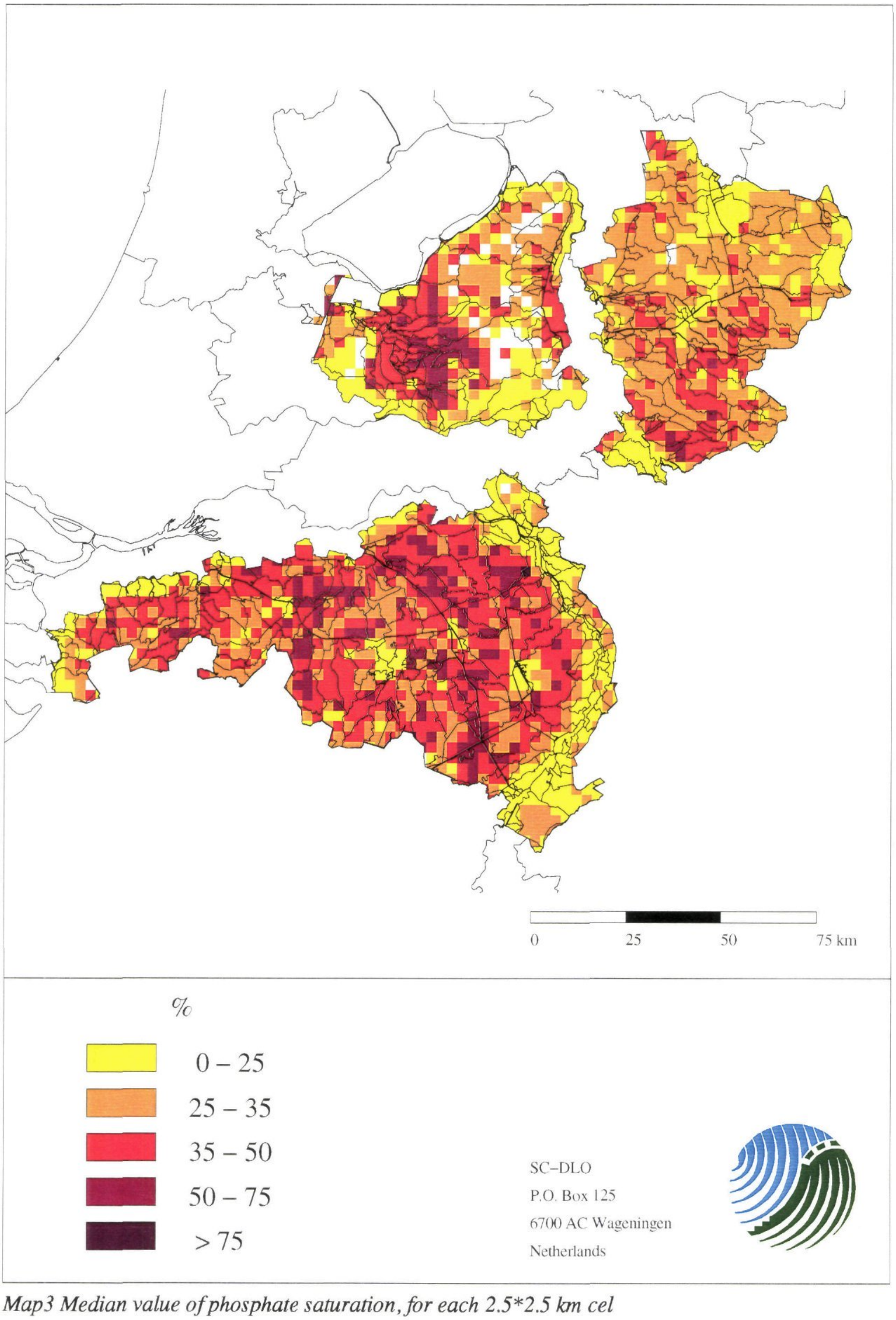
Plot P7: P leaching towards surfsce waters for the +10 kg (A) and +40 kg scenario (B)



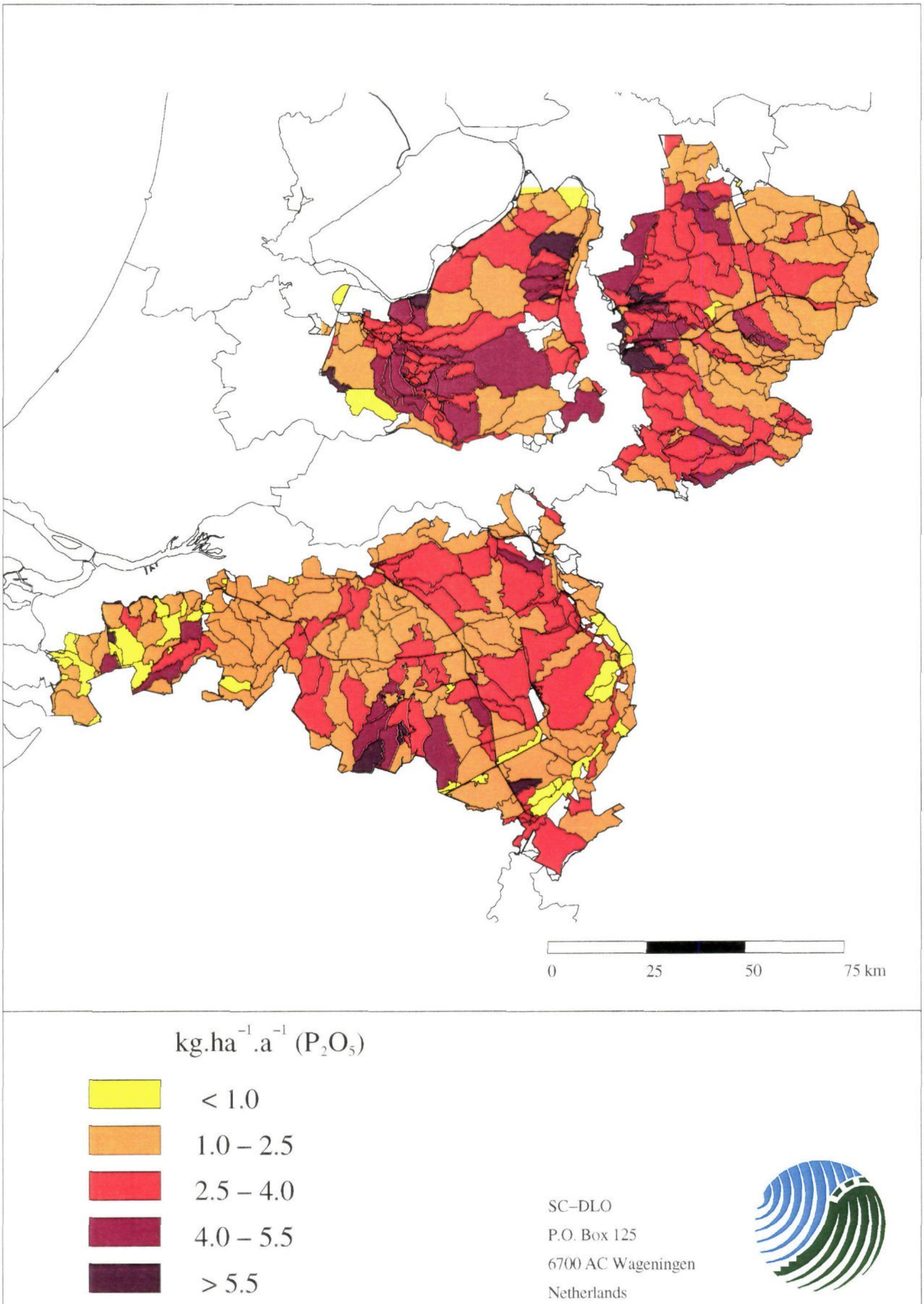
Plot P8: leaching fluxes at GHG-level for the +10 kg (A) and +40 kg scenario (B)



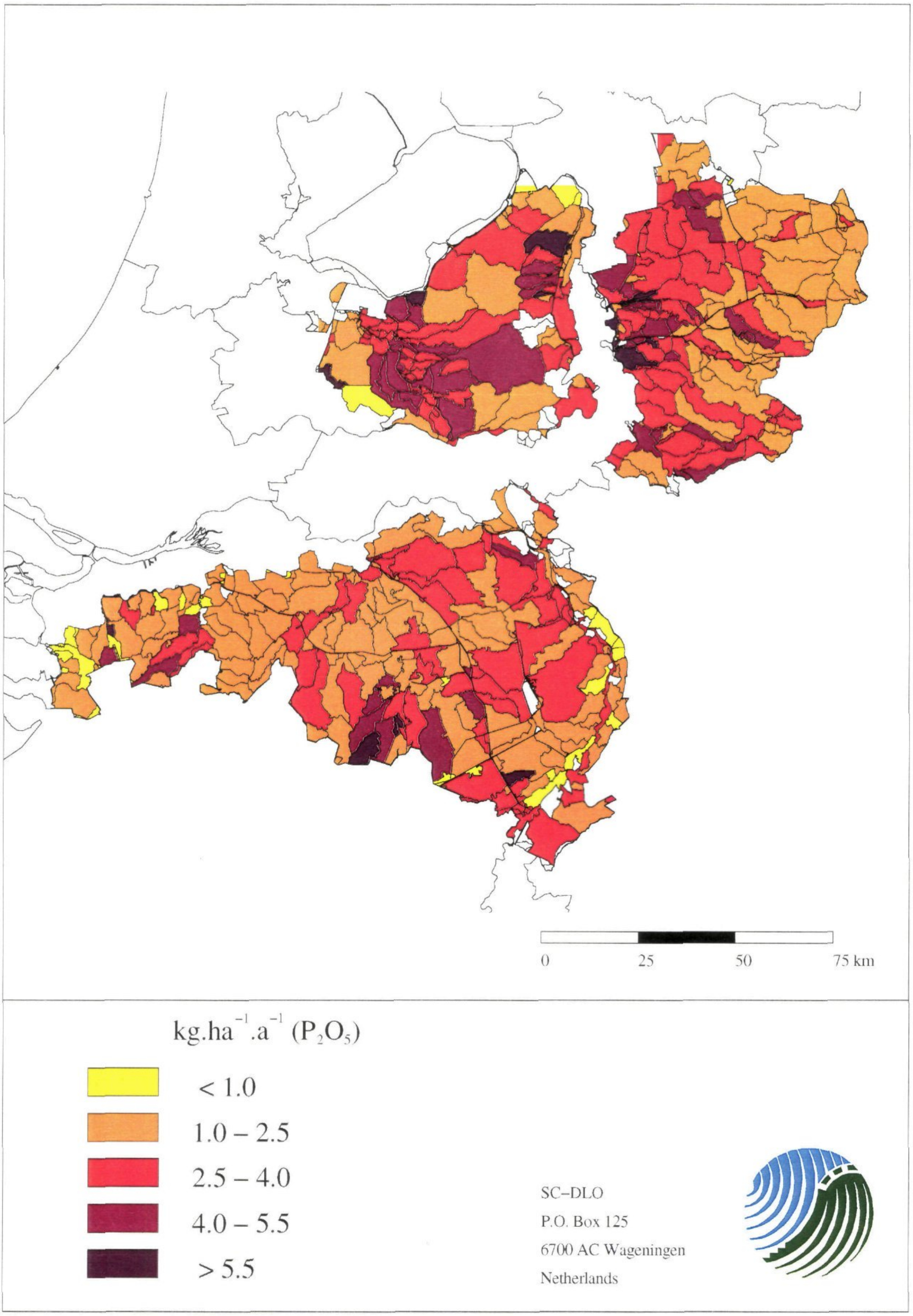
Plot P8: P leaching towards surface waters for the +10 kg (A) and +40 kg scenario (B)



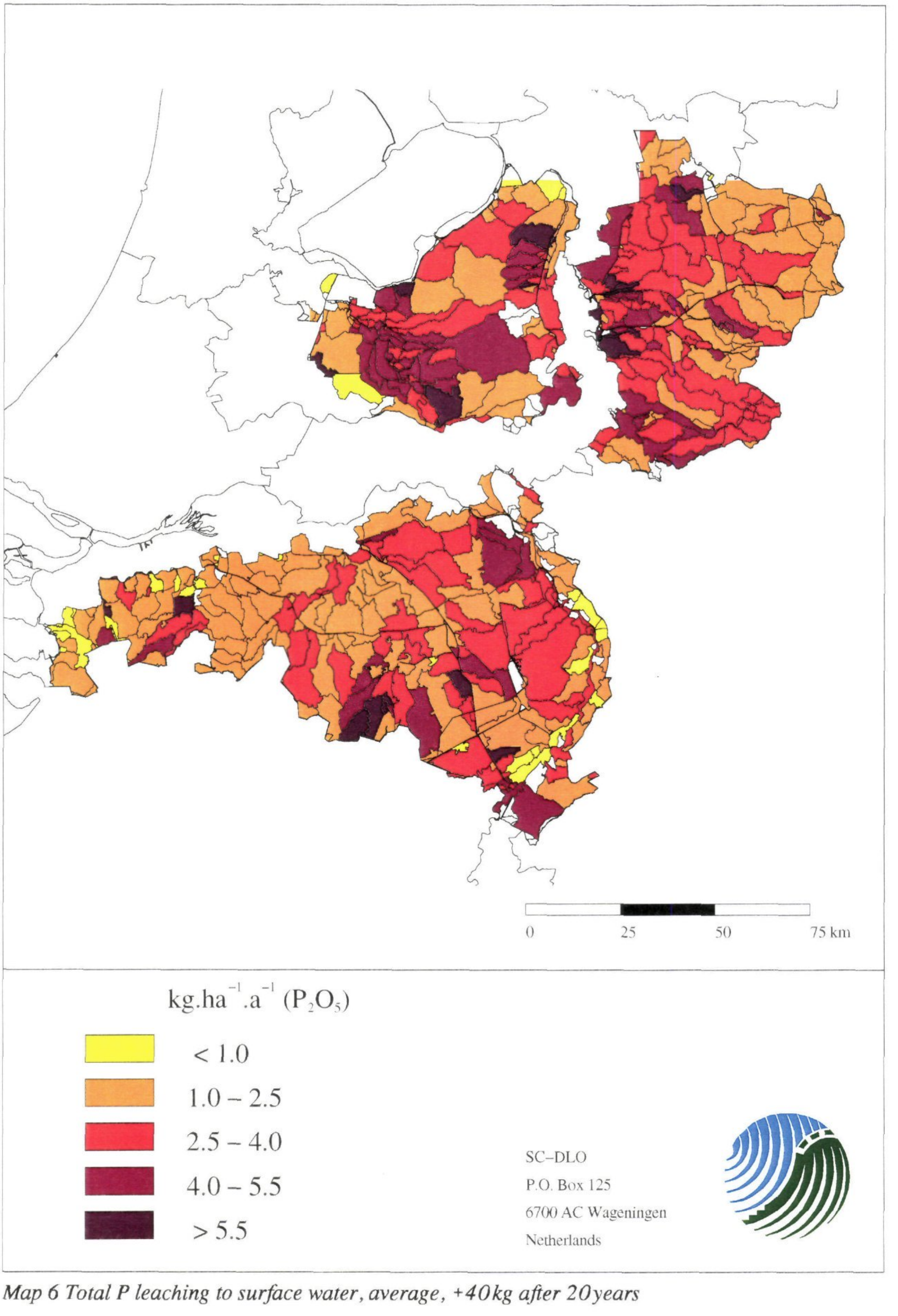
*Map3 Median value of phosphate saturation, for each 2.5*2.5 km cel*



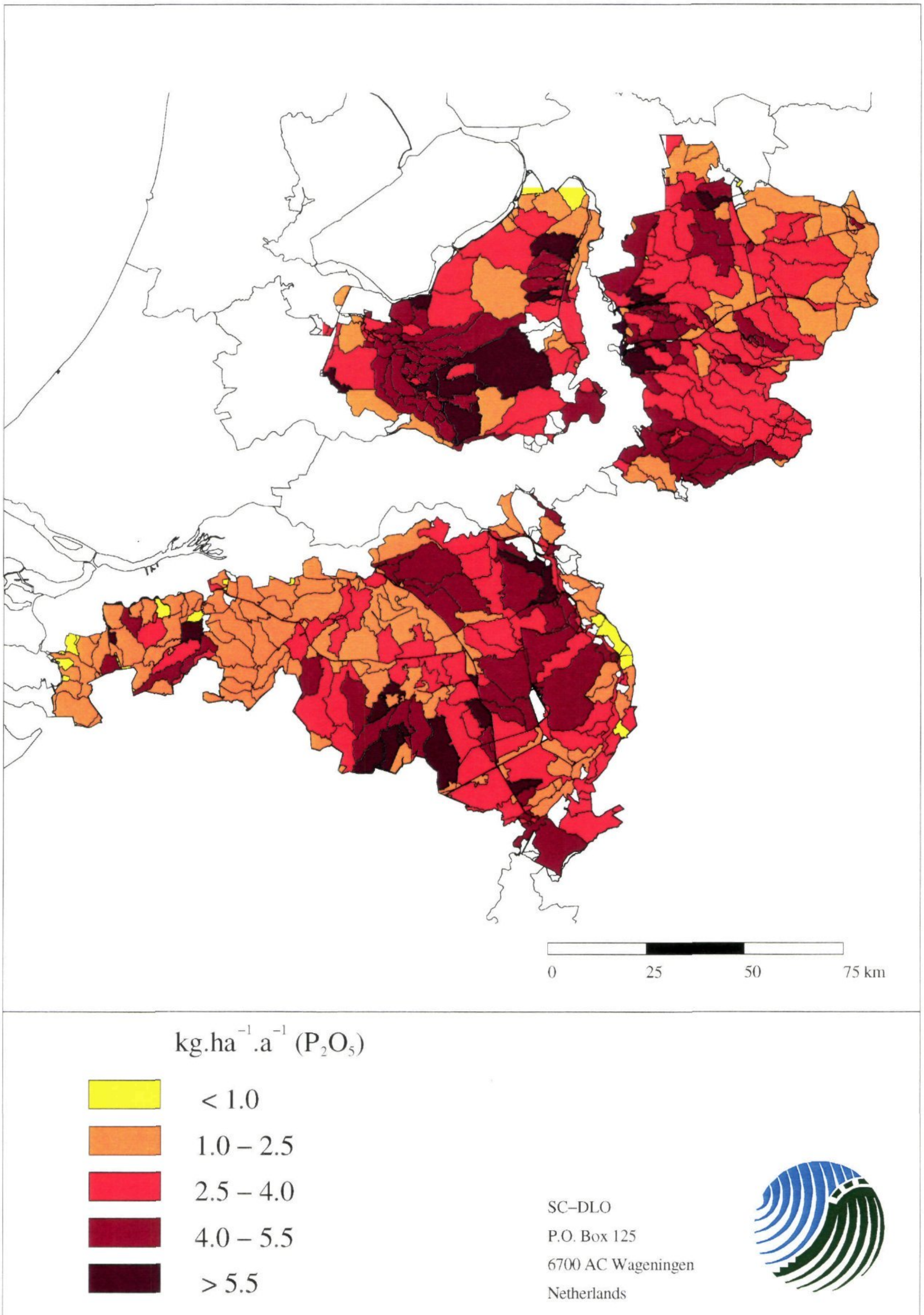
Map 4 Total P leaching to surface water, average, +10kg after 20years



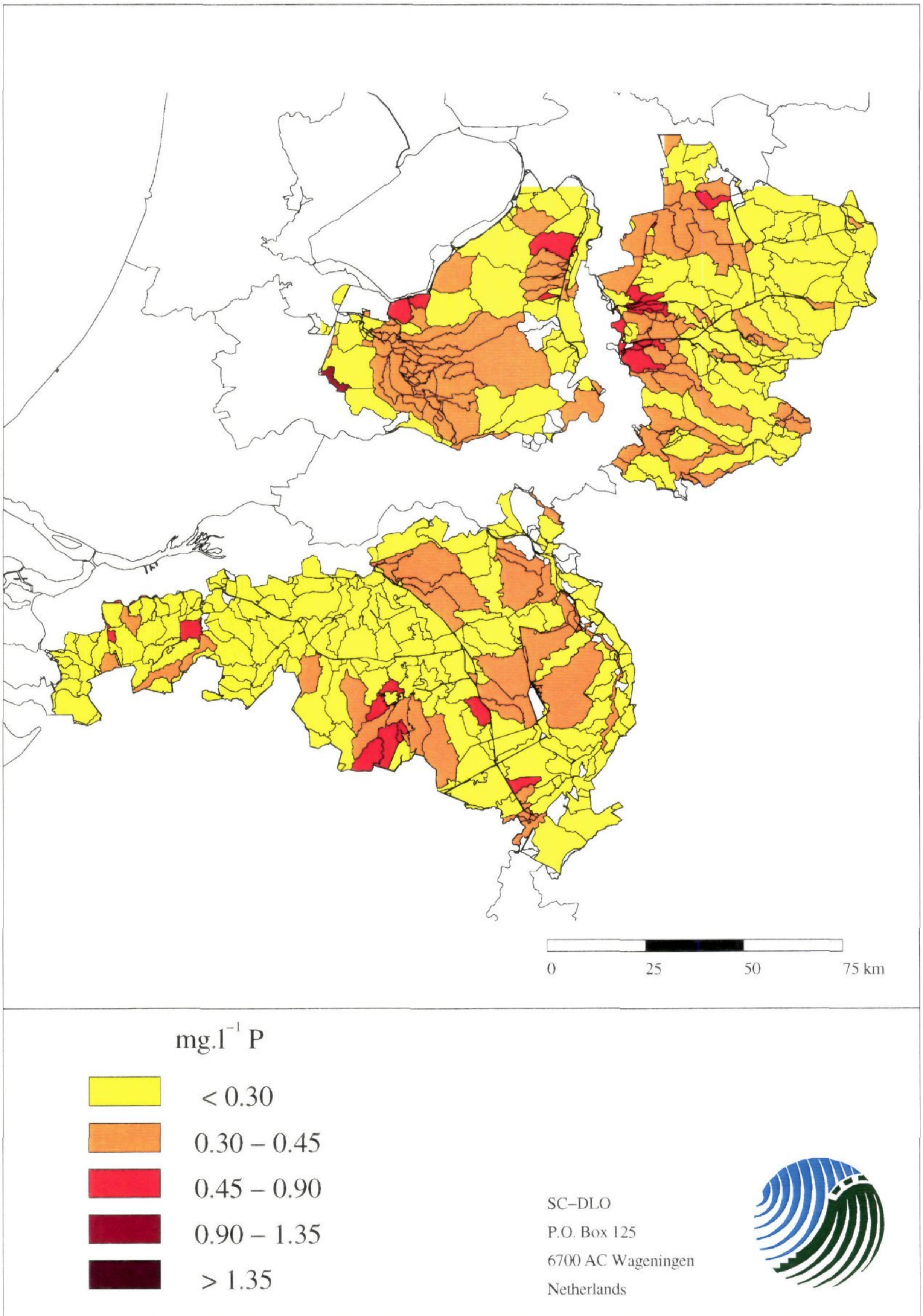
Map 5 Total P leaching to surface water, average, +10kg after 50years



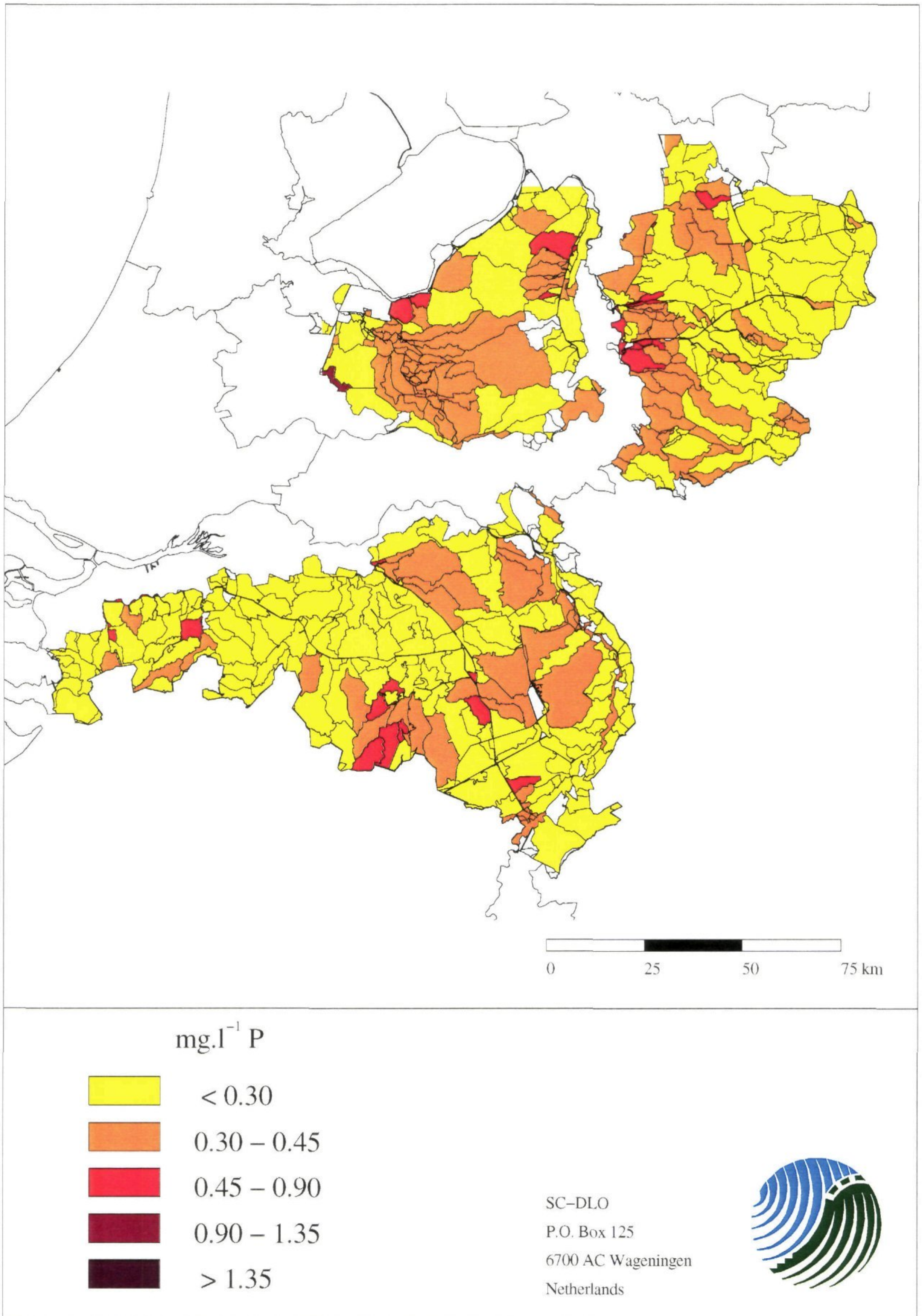
Map 6 Total P leaching to surface water, average, +40kg after 20years



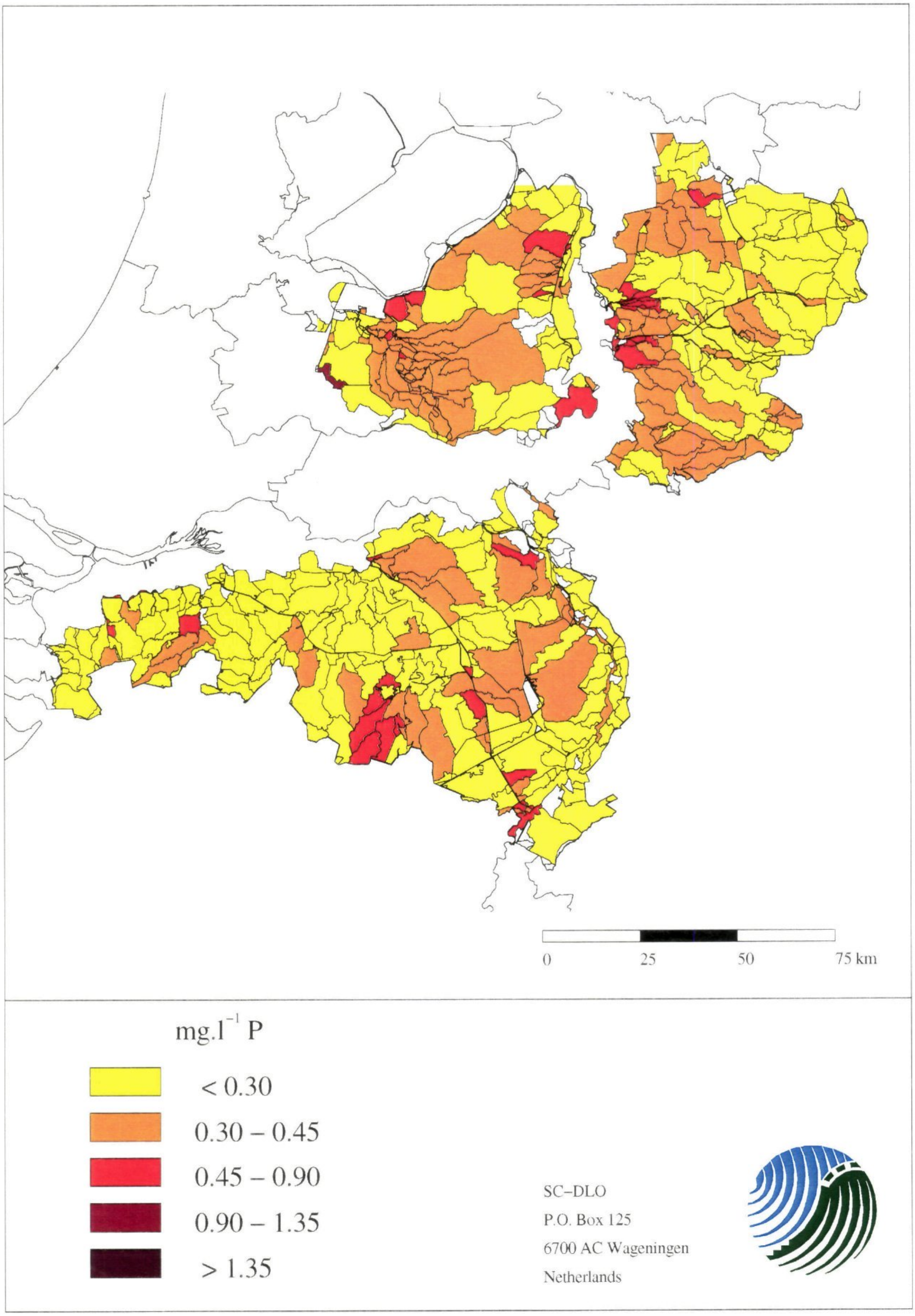
Map 7 Total P leaching to surface water, average, +40kg after 50years



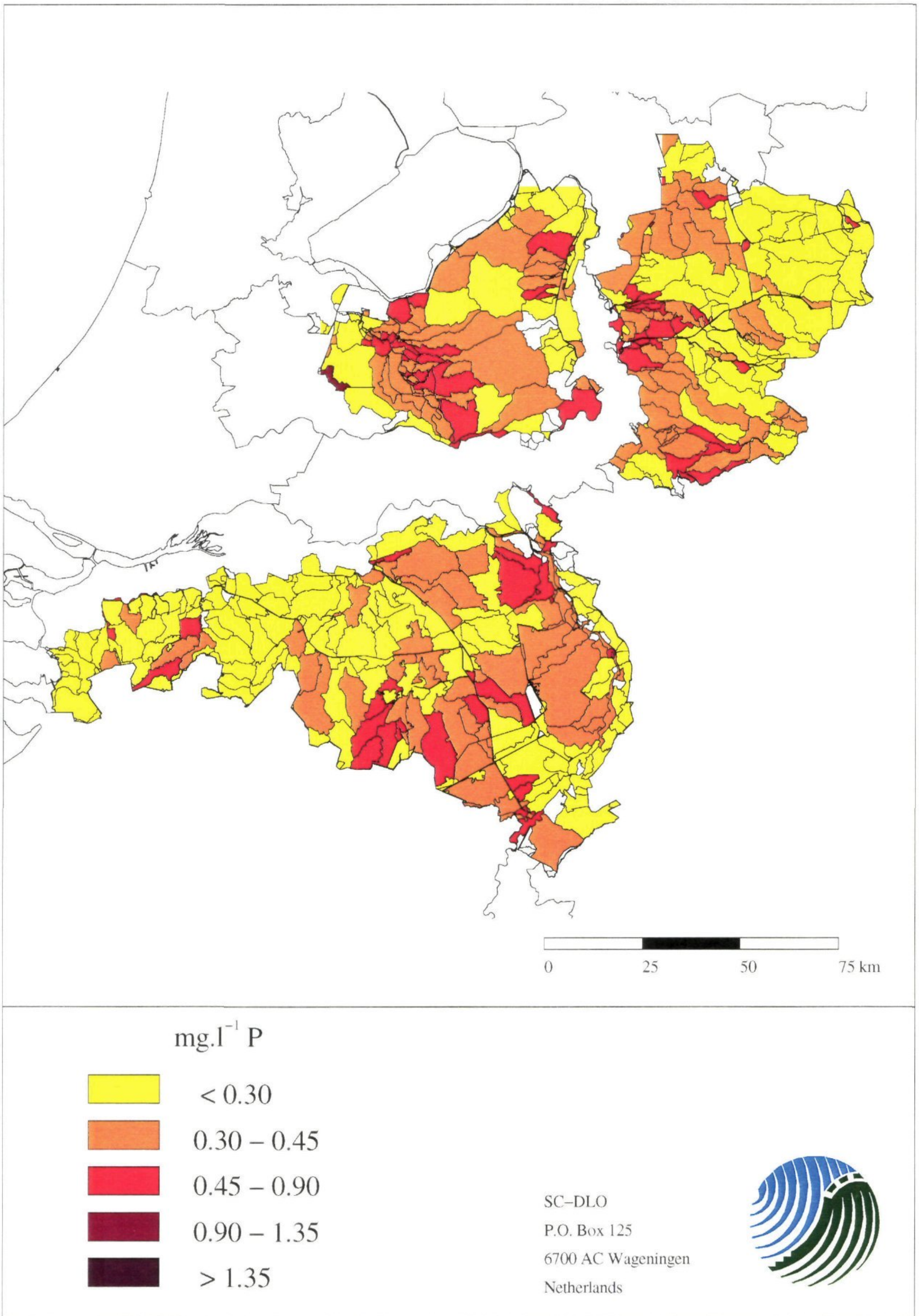
Map 8 Total P concentration in leachate, average, +10kg after 20years



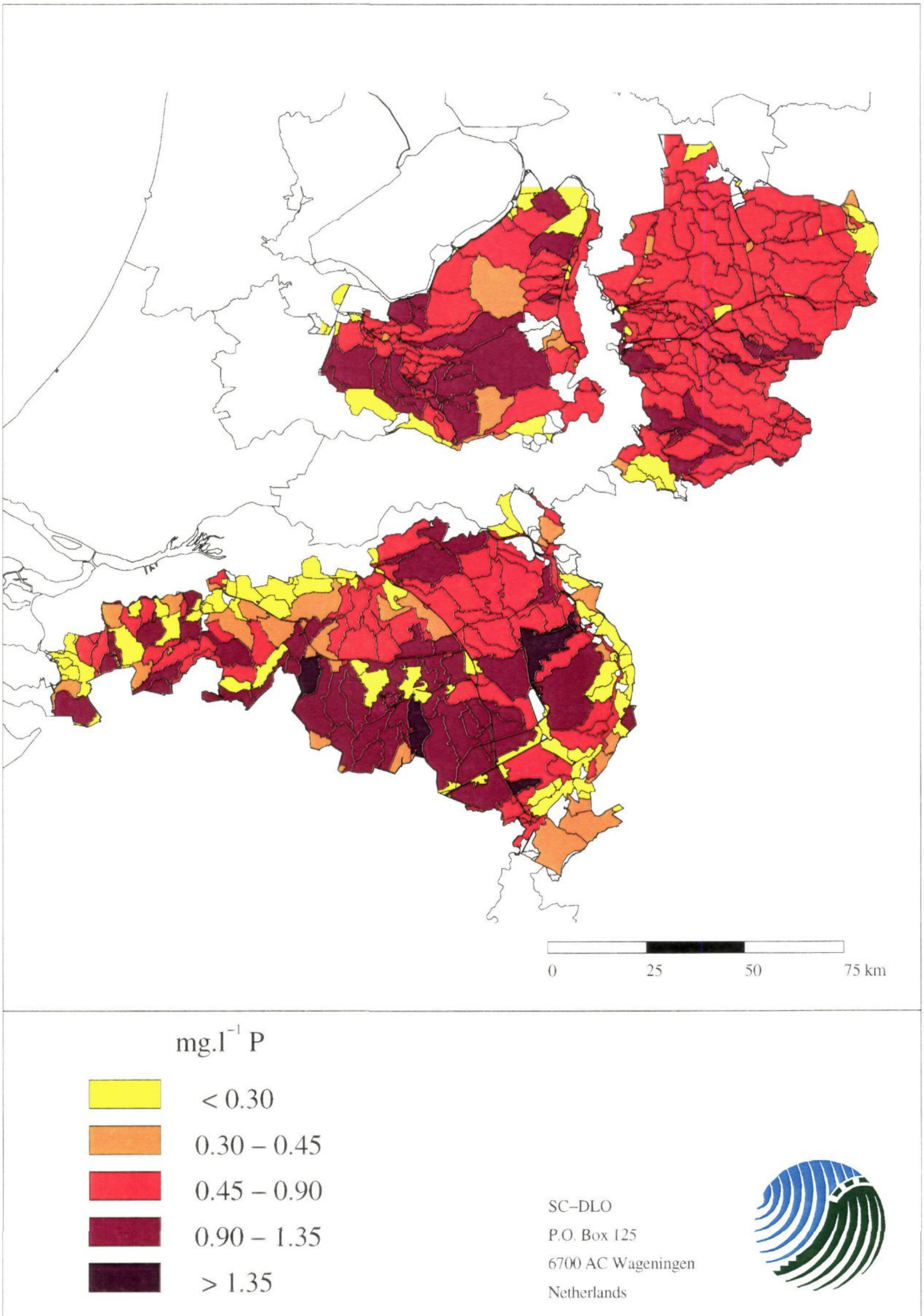
Map 9 Total P concentration in leachate, average, +10kg after 50 years



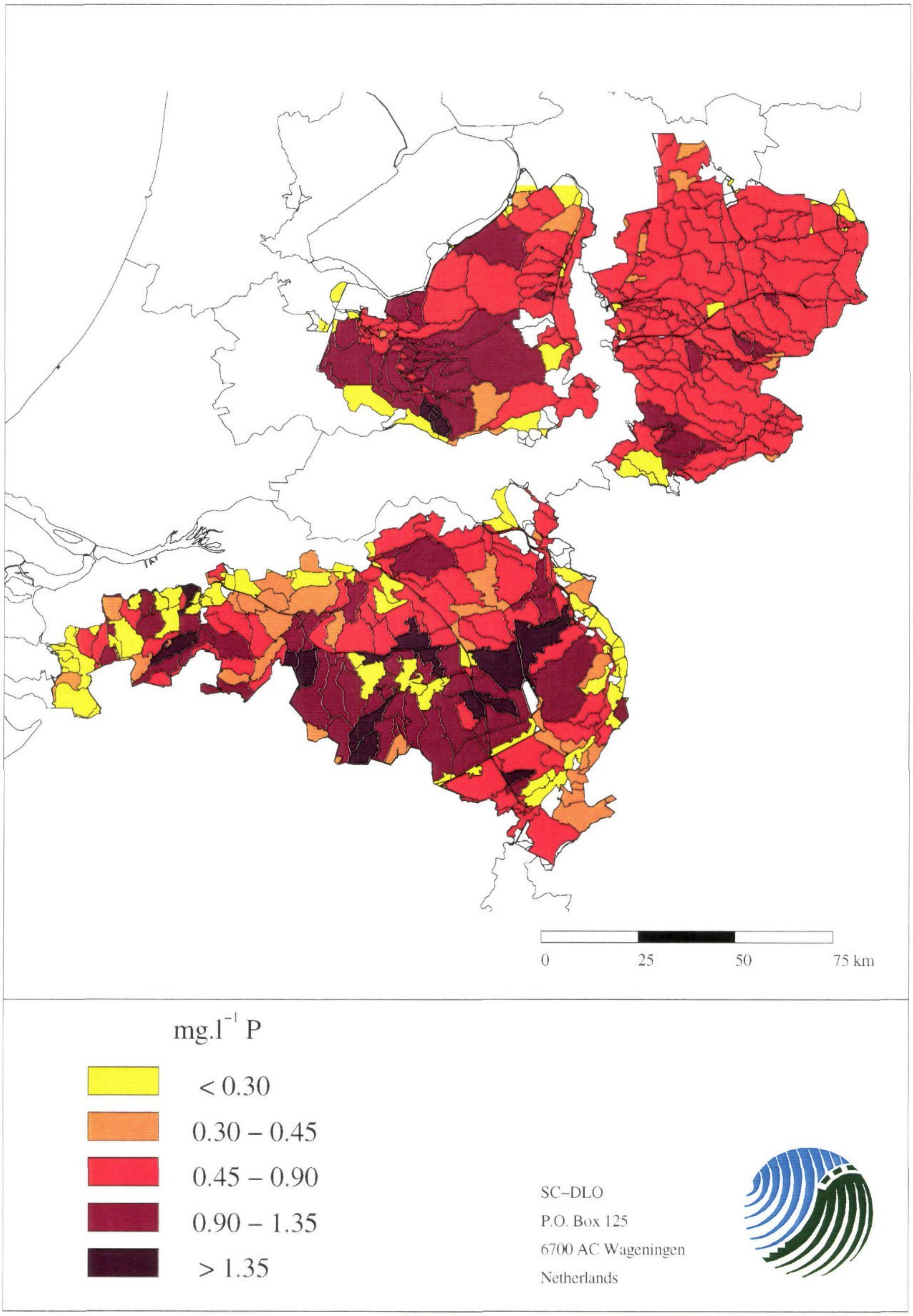
Map 10 Total P concentration in leachate, average, +40kg after 20 years



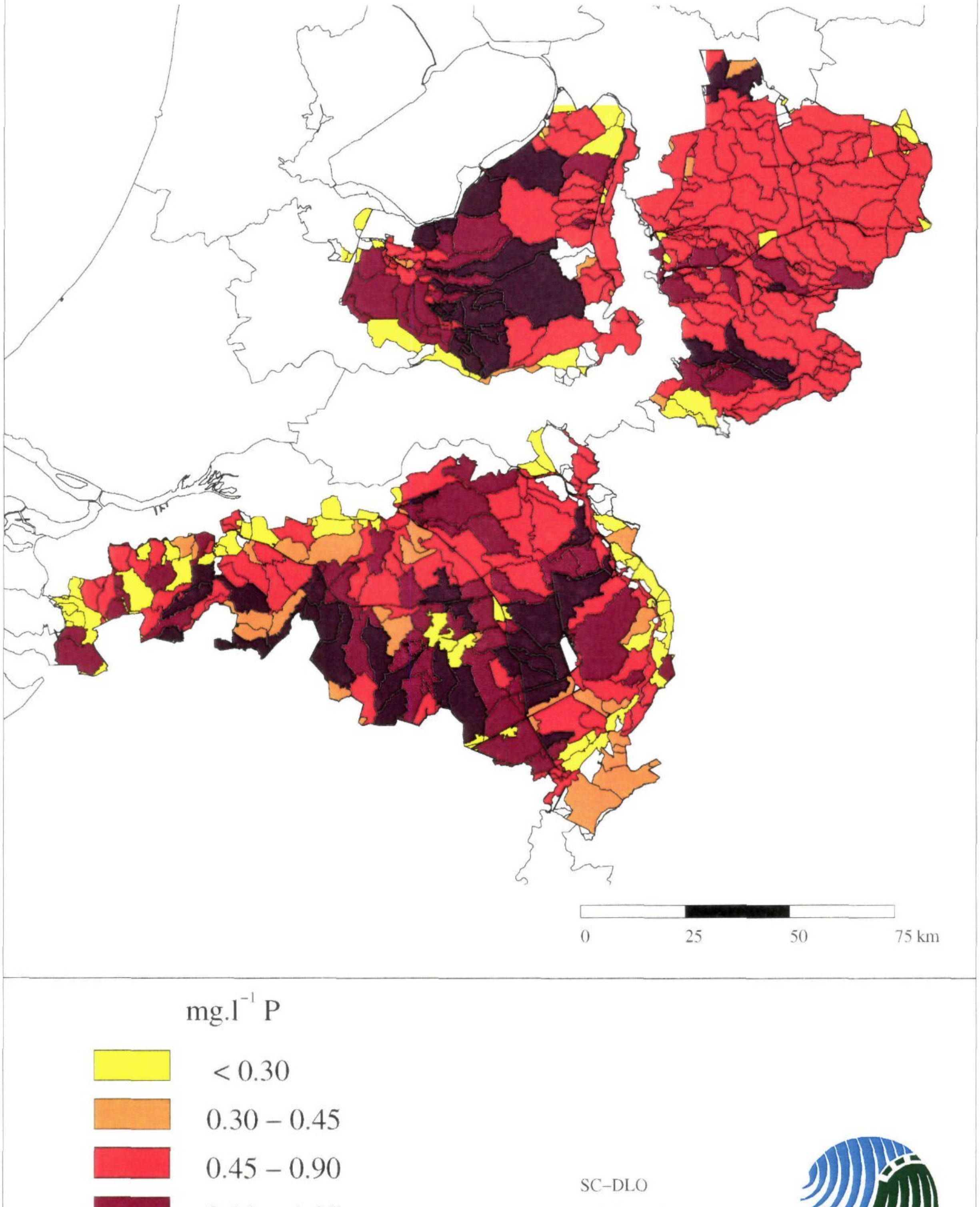
Map 11 Total P concentration in leachate, average, +40kg after 50years



Map 12 Total P concentration in leachate, 95 percentile, +10kg after 20years



Map 13 Total P concentration in leachate, 95 percentile, +10kg after 50years



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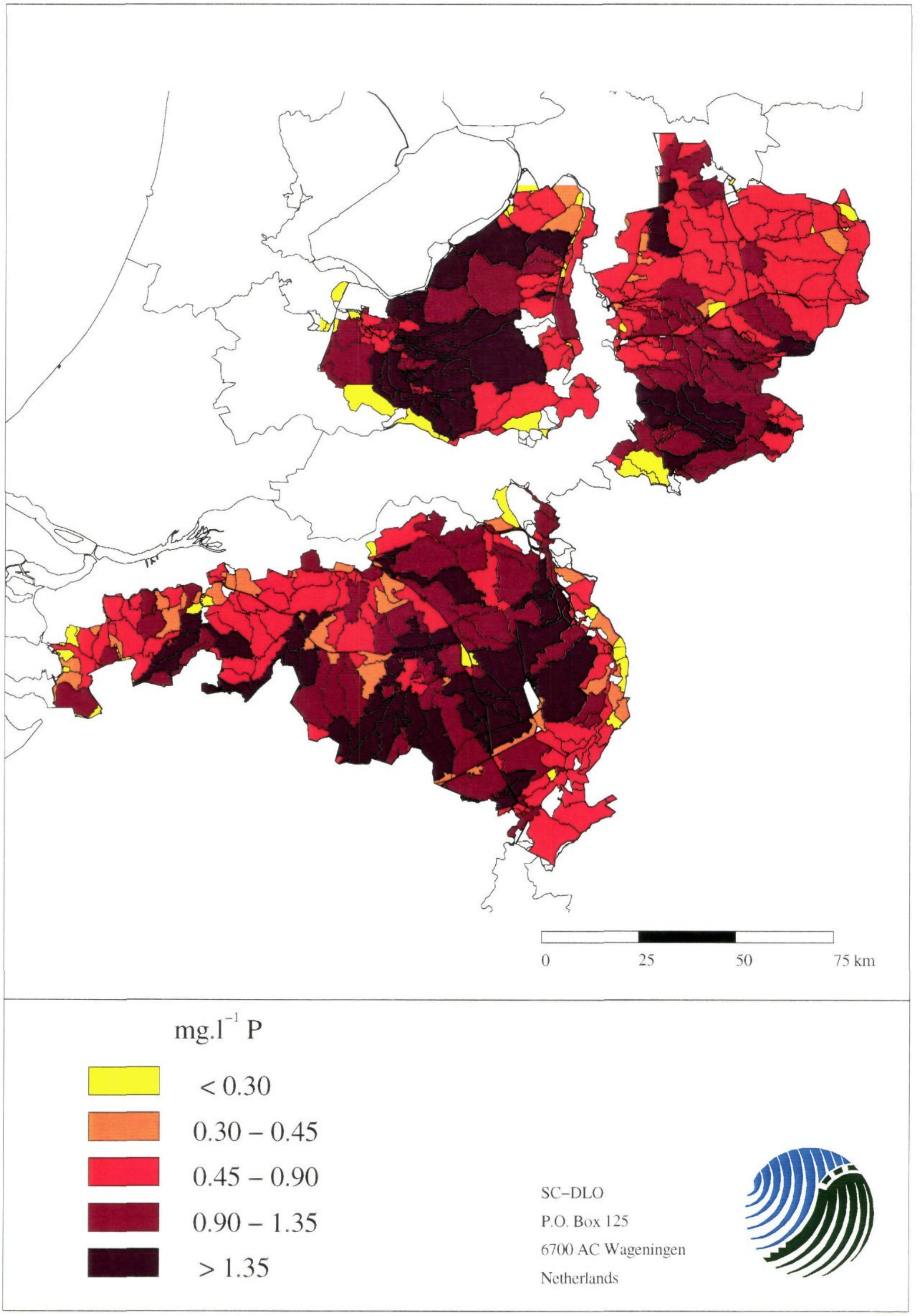
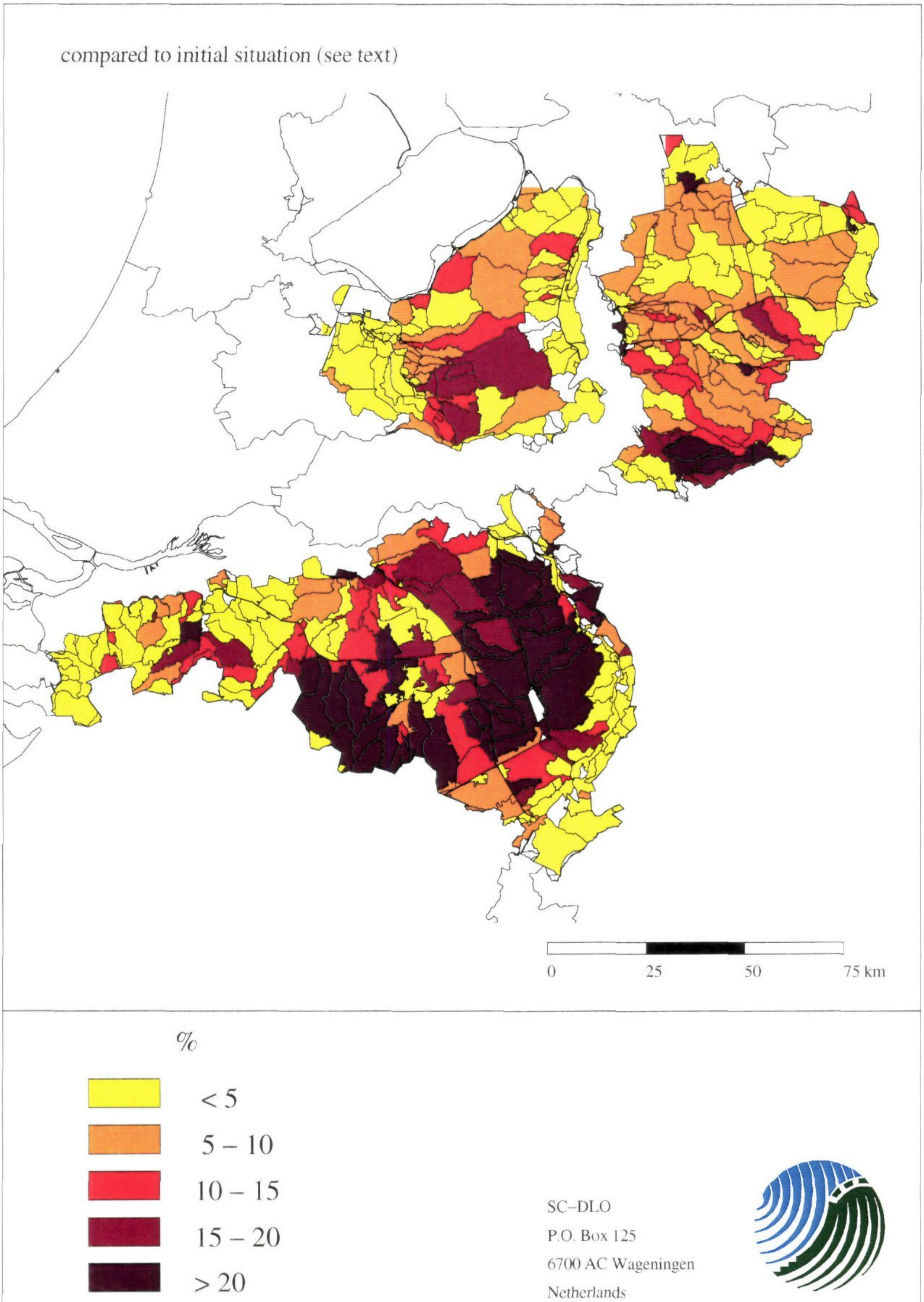
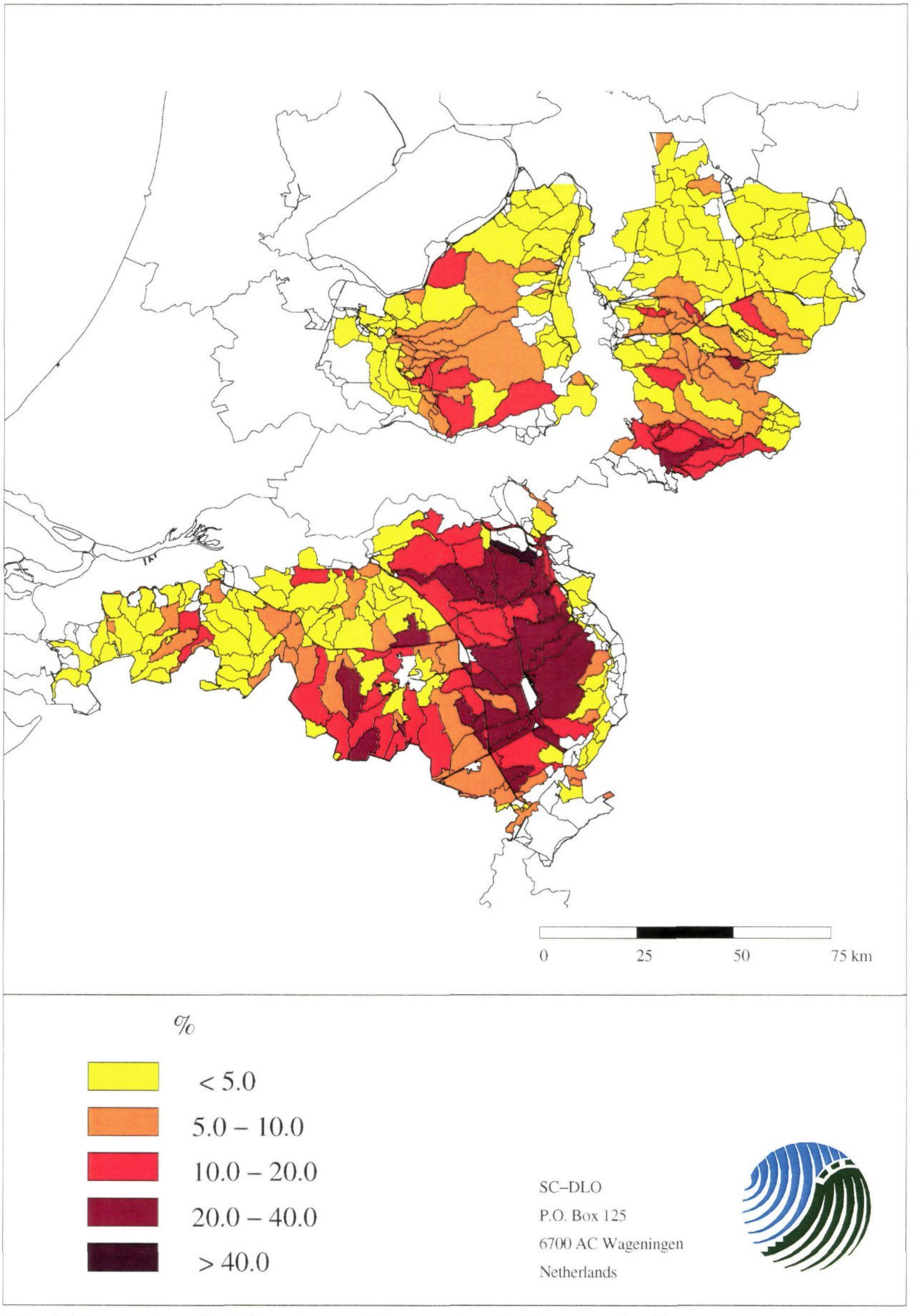
Map 14 Total P concentration in leachate, 95 percentile, +40kg after 20years



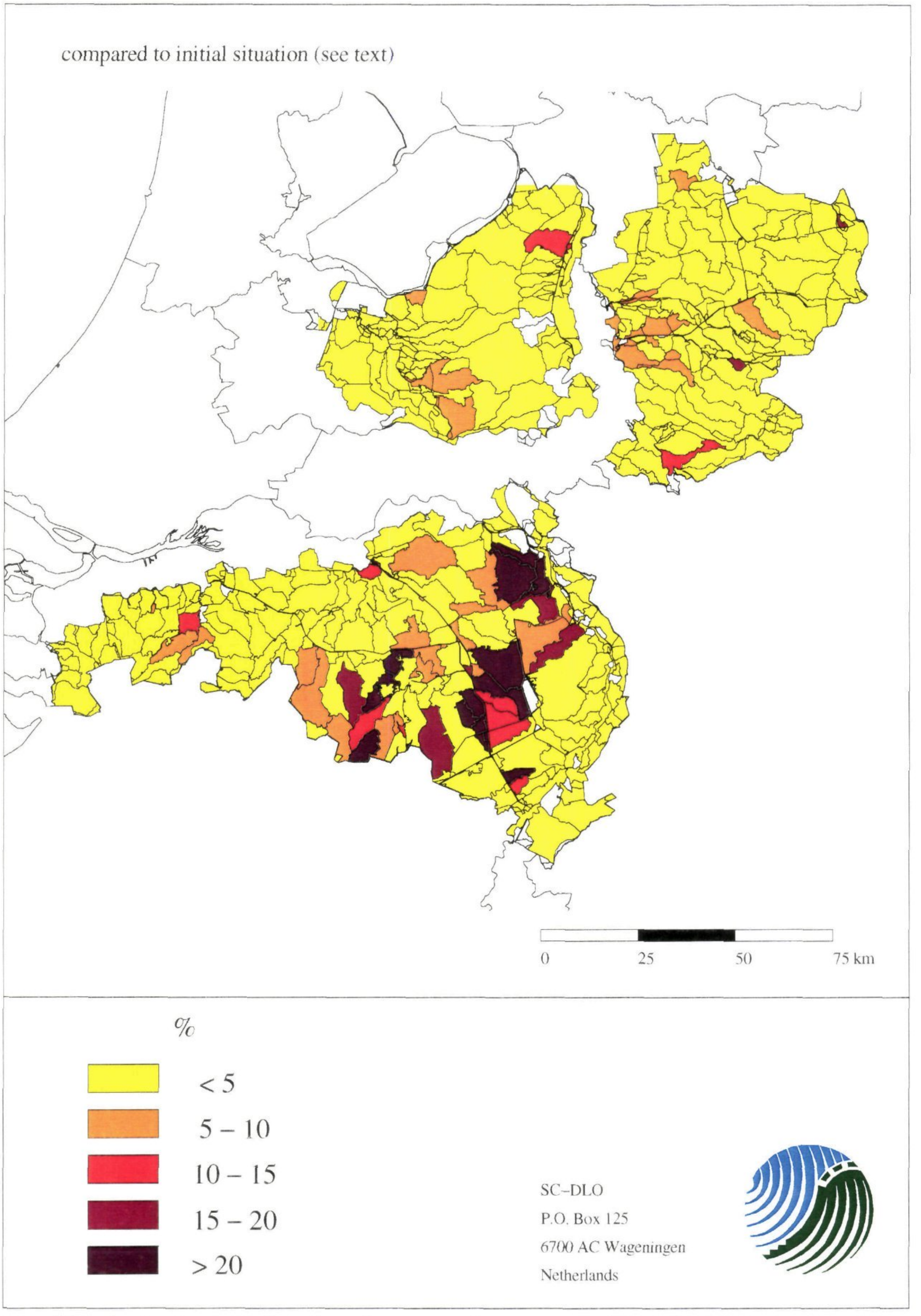
Map 15 Total P concentration in leachate, 95 percentile, +40kg after 50years



Map 16 Reduction in total P leaching after chemical treatment, + 10kg after 50years



Map 17 Percentage of catchment area with very strongly P saturated soils



Map 18 Reduction in total P leaching after chemical treatment, + 40kg after 50years