18 Soil as a waste management system
Restraints imposed by persistent constituents

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Summary

Soil can be considered as an extremely reactive biofilter in which processes of physical, chemical and biological nature take place. These processes may occur separately or in a combined way, simultaneously or subsequently. As such the soil system may offer an excellent opportunity for disposal and natural treatment of wastes, sometimes combined with efficient reuse of constituents present in the waste materials, for example in agricultural or horticultural crop production.

One of the main problems with wastes is brought forward by their composition. Even if they contain reusable constituents only, like in the case of potato starch wastewater, these usually do not prevail at optimum relative concentrations for fertilization purposes. If then application rates are adjusted to the constituent at minimum level, additional fertilization with for instance artificial fertilizers may allow optimal recycling of the waste without adverse environmental effects.

However, examples of such wastes are scarce. In most cases there is an imbalance of constituents in the wastes, especially if these contain compounds which cause potential hazards. Examples in this respect are persistent constituents like heavy metals or certain organic chemicals. Repeated application to soil of wastes containing such persistent hazardous compounds may introduce the build-up of a chemical time-bomb.

The same is true for phosphate. Repeated excessive application of this major plant nutrient element as compared to plant uptake occurs in areas with an animal manure excess problem. This causes saturation of the P-binding capacity of soil, which eventually will lead to phosphate leaching at unacceptable concentrations to groundwater and surface water. Once the leaching process starts, it will continue for very long time periods (decades).

Some important aspects of reactions in soil of major plant nutrients (N, P, K) from wastes are briefly discussed. Main attention is given to persistent constituents which impose limitations to waste usage in order to prevent pollution of soil, groundwater and surface water.

18.1 Introduction

Present society, especially in industrialized countries, produces huge amounts of waste products of all different types. In many cases soil is involved in the storage, disposal or reuse of these wastes. Sometimes the waste is of such a nature that only storage on or in soil can be considered, for example the inorganic part of household waste and car

wrecks. Many waste materials, however, do contain constituents which may be reused after application to soil. This contribution is confined to these types of wastes.

They include for example domestic waste water, waste water resulting from specific agricultural industries like canneries and potato starch processing, different types of compost, sewage sludge, and animal slurries. Because of the enormous expansion of animal husbandry in specific areas, animal manure is considered there sometimes more as a waste than as fertilizer.

It has been known for long that soil has the potential to 'treat' wastes in such a way that valuable constituents may be utilized at the same time, especially by means of plant growth and crop production. The oldest example, of course, is the use of 'night soil'. Far before the large scale introduction of treatment of domestic sewage water in waste water treatment plants, it was common practice in many communities to make use of so-called sewage farms. Here the soil in use for agricultural production (mainly as pasture) was regularly flooded with raw sewage water. By sampling and analyzing the resulting groundwater and drainage water it could be concluded that removal rates of constituents from the sewage water were extremely high. Depending on the type of soil it was found that phosphate removal was even higher than in tertiairy treatment plants (more than 98%, Beek & de Haan 1974). The soils involved were high in aluminum- and iron-(hydr)oxides. Of course it must be realized that those compounds which are not taken up by the crop or leached to the groundwater will accumulate in the soil and thus in the end may lead to soil saturation problems in such systems. Even domestic sewage water has the problem of containing not only organic matter and the plant nutrients N, P, and K, but also heavy metals are present. Moreover, disease germs can be spread in this way. Therefore cattle grazing on sewage farms had yearly to be vaccinated against splenic fever. This type of disadvantage, and especially the large surface area involved, caused that most sewage farms have been abandoned at present. The remainder from sewage water treatment in wastewater treatment plants, in the form of sewage sludge, still causes problems with respect to soil quality, when applied to land. This is obvious because these sludges contain potentially hazardous compounds in enriched concentrations as compared to the sewage water.

Specific wastewaters contain merely constituents of vegetables or agricultural products as in the case of cannery wastewater and potato starch wastewater. These wastes are in principle very suitable for reuse as fertilizers in crop production, as is for example shown by Campbell Soup Company in Texas, USA, and Scholtens Potato Starch Industry in Emlichheim, Germany. However, also here several disadvantages are involved as compared to the use of artificial fertilizers. In the first place one has to take into account the working coefficient of nitrogen. This working coefficient reflects the efficiency of a nutrient present in waste as compared to the use of the same nutrient in the form of artificial fertilizer. Because this working coefficient may vary considerably depending on climatic conditions during the growth of the crop, an important uncertainty results about the fertilization level. These coefficients always play a great part with the use of organic wastes for fertilization purposes, especially for nitrogen. For the other major plant nutrients, phosphorus and potassium, they are usually around 0.8-1.0, thus reflecting a more or less comparable efficiency as for artificial fertilizers. The latter, however, are available with different relative contents of N, P, K, thus enabling fairly precise adjustment of fertilizer application to soil fertility requirements. In wastewaters the relative contents of N, P and K are entirely dependent on the nature and composition of the products where the waste is derived from. Sometimes this composition is strongly deviating from fertilization requirements. In these cases the

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wastewater application rates should be based on the nutrient element which is at minimum level, with additional use of artificial fertilizer. But this is usually not done because of the extra costs and labour involved. In practice therefore wastewater application rates are usually taken so large that overdosing with certain nutrients results, which may induce various adverse environmental effects.

This imbalance of elements in most waste products constitutes the major problem with respect to their reuse. When this is true for the major elements N, P, K, it is even more so for elements or constituents like heavy metals, which are required for plant growth at relatively low level only. Sometimes these metals are abundant in wastes at any content because they are not necessary (and even hazardous) for proper soil functioning, as in the case of cadmium. The same may hold for persistent organic chemicals. It are especially such constituents which pose severe limitations on the reuse of wastes by application to soil because of their threat to soil quality. The wastes involved are for instance sewage sludges, certain types of compost, and so-called 'black soil' (an artificial mixture of wastes with peat and other soil material).

In the sequel some aspects of chemical reactions in soil of the major plant nutrient elements are first briefly discussed, especially in the light of possible adverse consequences for environmental quality. Then some attention is given to restricted use of wastes because of their undesirable composition.

18.2 Nitrogen reactions in soil

Main reactions of nitrogen in soil after application of waste (water) are schematically presented in Figure 18.1, taken from Lance (1972). A recent review on nitrogen reactions in forest soils was given by Johnson (1992) which also presents a complete analysis of the complexicity of nitrogen behaviour in soil. The main difference between forest and agricultural land is that nitrogen uptake by trees is relatively small as compared to agricultural crops. On the other hand loads are usually also small in forest as compared to agricultural production.

Figure 18.1 departs from the effluent of a secondary sewage water treatment (Flushing Meadows Project near Phoenix, Arizona), but comparable reaction mechanisms will apply when use is made of completely different waste materials like e.g. potato starch processing wastewater (de Haan et al. 1973), manure (Klausner et al. 1971), septic tank effluents (Walker et al. 1973), or generally any nitrogen source which does contain the nitrogen predominantly in the ammonium form. Taking the above picture as a guideline, the different reactions can best be discussed by tracing the various inorganic nitrogen compounds involved.

Under alkaline conditions part of the NH_4^+ entering the soil may revert to NH_3, and losses of N due to volatilization of NH_3 may then become substantial, as is extensively described in the contribution of van Breemen to this course about acidification of soil.

In order to prevent NH_3 volatilization after application of animal slurries to soil, an official regulation has been introduced in The Netherlands, which prescribes the practical way of application of the slurry. The slurry must either be injected into the soil or be worked in, e.g. by ploughing, directly after application. Deminishing NH_3 volatilization of course results in higher efficiency of nitrogen use, and hence larger problems with respect to possible nitrogen leaching. At present experiments are performed for acidification of the slurries by means of nitric acid addition. Although this may indeed prevent NH_3 volatilization, on the other hand the amount of nitrogen is then considerably enlarged which seems somewhat unlogical for areas where already a

great nitrogen surplus exists because of intensive animal husbandry. Acidification of the slurry requires renewed adjustment of the application rate of plant nutrient requirements. For situations where injection or working in of the slurry is impossible (peat soils) or undesirable (areas with pasture birds colonies) the new method may have some perspective.
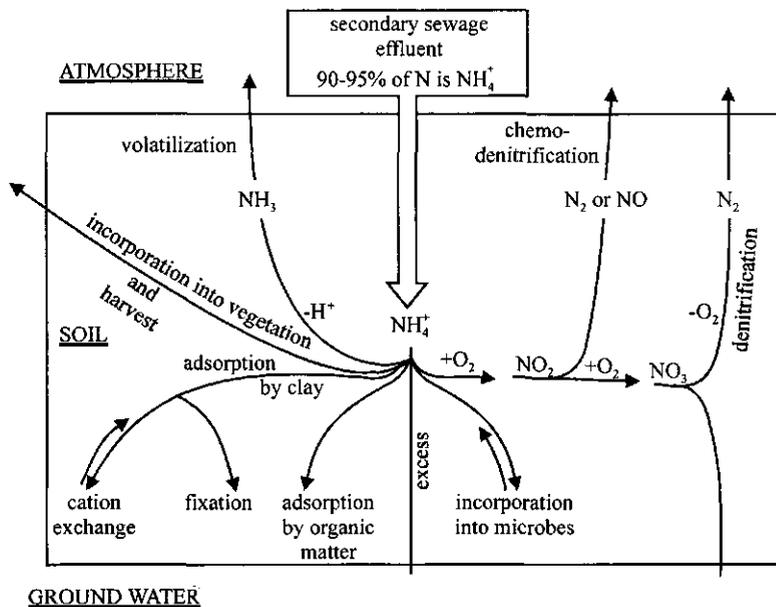
Figure 18.1. Transformations of nitrogen in soil, departing from a NH_4^+ -source.

A number of plant species may use ammonium as a primary source of nitrogen and hence part of the amount applied may leave the system this way. The remaining ammonium will now be displaced through soil subject to several types of adsorption and fixation processes, namely regular cation exchange, intra-lattice fixation much alike K-fixation as is described later, and direct chemical incorporation into some humus intermediates. Finally ammonium may become incorporated into microbial tissue, thus re-entering living organisms, and again be subjected to mineralization upon a decrease of the size of the microbial population.

Ammonium not intercepted through one of the above mentioned mechanisms may leach to deeper soil layers and eventually reach the groundwater. It may, however, also undergo nitrification if the specific requirements for this reaction are met.

The main conditions for nitrification are:
- sufficient amounts of nitrifying micro-organisms;
- sufficient supply of free oxygen;
- presence of suitable energy sources;
- satisfactory conditions with respect to moisture and temperature levels.

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The nitrification process itself consists of a number of different steps in which intermediate compounds are formed. Formation of several of these NO_x compounds is undesirable as they may, after volatilization, contribute to destruction of the ozon layer.

Nitrate ions formed by nitrification or added as such to the soil are highly mobile in most soils because they are not significantly adsorbed onto the soil solid phase. They are thus leached down with the carrier stream (the retardation factor equals zero), unless they are intercepted by plant roots and taken up as nutrient ions. Nitrate leaching may result in undesirable quality of groundwater to be used for drinking water purposes because of giving use to methaemoglobenimia ('blue baby disease'). Therefore nitrate drinking water standards are set by the WHO at 50 mg NO_3 per liter; the EC has lowered this value to 25 mg NO_3 per liter.

Figure 18.2, resulting from groundwater analysis in an area with large manure surpluses, shows that here these standards can not be met anymore. It must be feared that the increase of nitrate contents, as measured over the past decades, will still continue, even if any further nitrogen application to soil would be terminated from now on, because of excessive manure applications in the past.
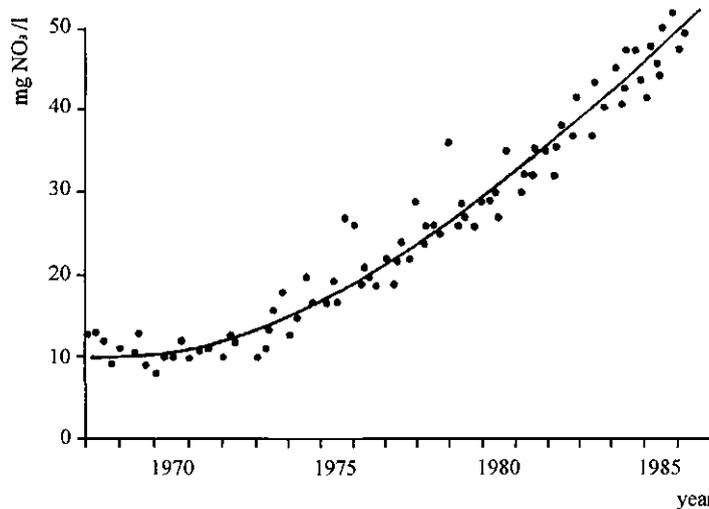

Figure 18.2. Nitrate content in deep groundwater, as a function of time, in an area of intensive animal husbandry.

The only way of removal of nitrogen from the system, once present as nitrate, is given by the microbial process of denitrification. In contrast to nitrification this process requires reducing conditions, plus again an energy source. Accordingly these processes, if occurring both in one system, will occur in a sequence in time and/or in space. As furthermore denitrification, leading to the formation of gaseous N_2, acts only on nitrate ions, nitrification must preceed denitrification if nitrogen enters the soil as ammonium. In a global sense, denitrification constitutes the closing link in the nitrogen cycle, since only in this way nitrogen returns to its ultimate source, namely N_2 of the earth's atmosphere.

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Summarizing the situation with respect to the management of excess of nitrogen compounds entering locally the soil as in many waste management systems, it may be concluded that undesirable environmental effects are to be expected primarily from NH_3 volatilization, and from excesses of nitrate ions leaving the topsoil and reaching groundwater or open water. Although the soil is capable of retaining temporarily increased levels of nitrogen, this is of no avail on the long run as such retention implies a labil storage in organic nitrogen forms. In practice such storage will eventually lead to a new steady state at an increased intensity level where the larger input is accompanied by an increased output of the mineralization and leaching processes, exerting an undesirable impact on the environment. Only if the increase of the input is balanced by a removal via harvested crops or by increased denitrification will it be possible to maintain the output via leaching water at an acceptable level.

18.3 Phosphate reactions in soil

In most wastes P prevails in the form of inorganic phosphates. This is also the case with animal manures, where roughly 80% is found in inorganic form (Gerritse 1976). Soils usually have a very large bonding capacity for phosphate. For detailed information about the reaction mechanisms involved the reader is referred to the contribution of van Riemsdijk and van der Zee to this course, and the extensive literature reviews presented there. The attention will be restricted here to the practical effects of this phosphate bonding in case of waste application to soil, and the environmental consequences of phosphate saturation of soil systems.

Phosphate in soil is largely immobile, being either adsorbed or precipitated, while a small fraction may move with leaching water at concentrations which are generally not larger than about 10^{-5} molar. Reports on phosphate distributions with depths thus invariably indicate an accumulation in the topsoil layer. This was even the case in systems of very high phosphate applications as on sewage farms (Beek & de Haan 1974). Figure 18.3 shows the course of the P content with depth, expressed as mg P_2O_5 per 100 g of dry soil, on a sewage farm, which is in normal agricultural use as grassland for pasture. The fields are, already for more than 60 years, flooded on rotation basis with raw domestic sewage water at a rate of about 350-400 mm each month.

The dotted line in Figure 18.3 represents the situation on non-flooded soil. To this purpose soil profiles were sampled on the forest land soils that surround the farm and from which the sewage farm has been reclaimed. The difference between both situations refers to a total amount of 15,700 kg P_2O_5 per hectare; 90% of this total phosphate storage is obtained in the upper 40 cm and even 65% in the 20 top layer (which is regularly ploughed). One hundred percent efficiency in phosphate removal from the water would lead to a so-called block-front in the proceeding of P-saturation of the soil. Some dispersion always occurs because of heterogeneity of the soil system.

This soil property of effectively bonding of phosphate ions has thus been applied in practice for removal of phosphate from wastewater. At the same time it is the reason that excessive supply of phosphate in manure slurries in certain areas has not led so far to large-scale leaching of phosphate from soil, which would damage surface water bodies by eutrophication. On long term, however, any significant overdose as compared to plant uptake will inevitably induce complete saturation of the bonding capacity. Again of course, the time scale involved is depending on the magnitude of the remaining bonding capacity on the one side and of the overdose on the other.
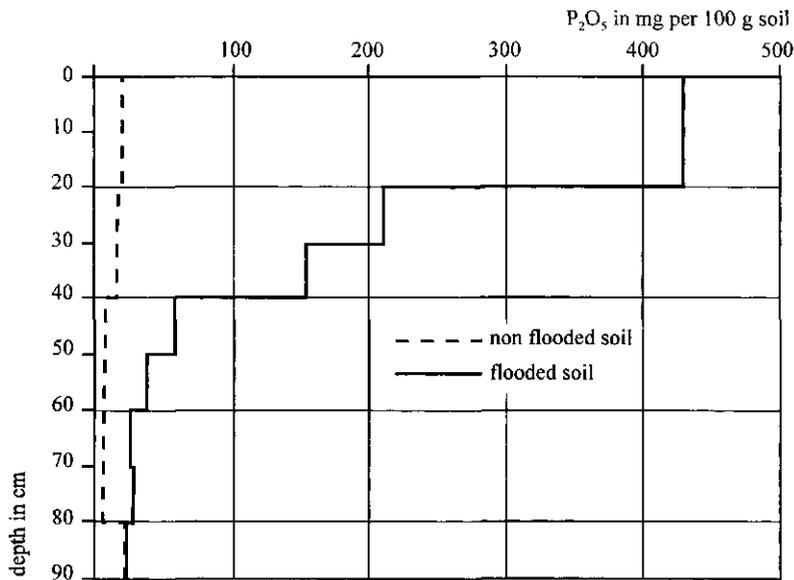
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Figure 18.3. Phosphate distribution with depth on a sewage farm as compared to the non-treated soil.

Behaviour of phosphate in soil thus received considerable attention in research programmes. Initially mainly with respect to plant nutrition and soil fertility aspects presently also in relation to possible environmental damage. Main soil compounds responsible for phosphate bonding are organic matter, clay minerals and (hydr)oxides of Al and Fe. Beek & van Riemsdijk (1979) gave an extensive review on phosphate bonding in soil. A model which satisfactorily describes the bonding mechanism in sandy soils was developed by van Riemsdijk et al. (1984) and van der Zee & van Riemsdijk (1986; 1987). A formula of general validity for homogeneous soil describing the transport of the phosphate saturation front in soil was presented by Lexmond et al. (1982). As long as the shape of the saturation front in penetrating the soil does not alter, the rate of movement of this saturation front may be described with the following simple relationship between the proceeding saturation, phosphate load and bonding capacity:

$$y = \frac{(A - U)}{B.S} \cdot 141 \tag{1}$$

in which

y = velocity of saturation front, in $cm.yr^{-1}$
A = phosphate supply to soil, in kg P_2O_5 $ha^{-1}.yr^{-1}$
U = uptake and removal by crop, in kg P_2O_5 $ha^{-1}.y^{-1}$
B = dry bulk density of soil, in $kg.m^{-3}$
S = remaining phosphate sorption capacity, in mmol $P.kg^{-1}$.

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The value of S thus constitutes a crucial point. It may be determined by the use of sophisticated techniques as developed by van Riemsdijk et al. (1986). Fortunately it can also be reasonably well approached by analytical techniques which are more easily accessible. As shown by Lexmond et al. (1982) and van der Zee (1987) for many soils a good correlation exists between the total amount of P that can be bound and the amount of reactive Fe and Al as determined by oxalate extraction. However, it is the presently remaining bonding capacity, S, that counts. S thus is the main parameter for vulnerability of soil for phosphate leaching. The relationship of equation (1) has graphically been visualized in Figure 18.4.
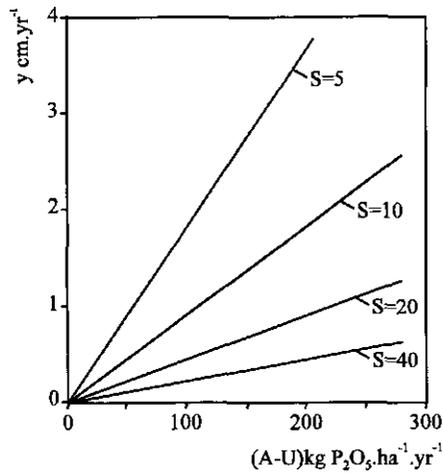

Figure 18.4. Velocity of phosphate saturation front in soil as a function of net P addition, for a number of values of remaining phosphate bonding capacity; soil bulk density = 1500 kg.m^{-3}.

Table 18.1. P removal from soil, in kg P_2O_5 ha^{-1}.yr^{-1}, for a number of crops at assumed dry matter production corresponding to good yield.

Crop	Dry matter production kg.ha^{-1}	P removal kg P_2O_5 ha^{-1}
wheat	9,000	57
rye	7,200	32
corn (grain	4,700	32
silage corn	12,500	82
potatoes	9,200	55
sugar beets	19,600	102
hay	4,000	32

The value of U is related to the type of crop and to crop yield. In Table 18.1 some examples are presented. On pasture farms P removal is usually very low because animal products like meat and milk are the only ways of discharge there.

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The value of A (P supply with fertilizer) is strongly dependent on the fertility status of soil with respect to phosphate, which again is inversely related to the remaining bonding of P in soil. There is not much sense in giving average data for this supply as can be calculated from P fertilizer consumption in the different EC member states and agricultural areas involved. Agricultural soils with extremely low capacity for phosphate bonding will probably first induce the P leaching problem. Probability for saturation of soil with P by the use of mineral fertilizers is low when the official fertilization recommendation of extension services are followed. Then the supply is adjusted on the basis of P availability according to soil sample analysis. The availability will automatically increase with proceeding saturation of the bonding capacity, and when saturation is reached, the recommended supply should not exceed P removal by crop. There are several reasons in practice, however, for deviations from these official recommendations. One would be non-regular soil sampling and analysis, and supply on the basis of farmers' experience. Another one would be the use of *e.g.* 'Starting' doses of mineral P in silage corn, even when the P fertility status of soil is very high. As compared to mineral P, however, P overdosing as the result of manure surpluses seems far more important since in that case values of (A-U) may easily amount to several hundreds of kg P_2O_5 $ha^{-1}.yr^{-1}$.

The approach presented above also allows a risk evaluation for P leaching. Figure 18.5, taken from Van Riemsdijk et al. (1983) presents an example.

fraction

0.3

0.2

0.1

0.0

N = 165

0   2   4   6   8   10   S-r/(ton P_2O_5)

0   20   40   60   80   100   I
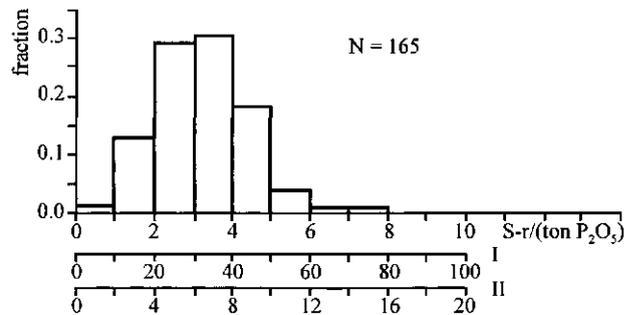
0   4   8   12   16   20   II

Figure 18.5. Frequency distribution of remaining P sorption capacity for topsoil of 165 different parcels in a certain region in the Netherlands. Axes I and II refer to time of saturation (years) for P supply rates of 160 and 560 kg P_2O_5 $ha^{-1}.yr^{-1}$.

Here the frequency distribution for the value of the remaining P sorption capacity, expressed in metric ton P_2O_5 ha^{-1}, has been given for the top layer of 165 agricultural parcels in a certain region in The Netherlands. By translating the capacity axis for given supply rates (160 and 560 kg P_2O_5 $ha^{-1}.yr^{-1}$, in this case), a transformation is obtained to the number of years it will last before this layer will be saturated with P.

Saturation of soil profiles should be avoided because the combined effect of desorption and continued supply will harm surface water quality. These effects are most severe with animal manures, because of the high P concentration involved. In that case the phosphate concentration in the leachate exceeds the eutrophication standard with a factor of about thousand indicating that 1 m^2 of satured soil may ruin the water quality of 1000 surrounding m^2.

It is therefore that in the Dutch General Administrative Order (GAO) on manure application special attention is given to the phosphate saturation problem.

In order to prevent soils from phosphate leaching, the decree on use of animal manure applies standards based on the phosphate content of the manure. A differentiation was made between application rates for grassland, silage corn and arable land. The final goal is set at an application rate which balances the uptake of phosphate by crops, which is on the average 70 kg P_2O_5 per hectare per year. This is in accordance with the principle that addition of substances to soil should not exceed the (acceptable) removal by crops and by leaching. Because of the excessive availability of manures, this goal can be reached only after a certain time period, during which the application rates are stepwise decreased. Table 18.2 gives the application rates for the various soil uses and different phases. An exception clause was introduced for those soils which are to be considered as phosphate saturated. Here, the final rate was made applicable from the moment of effectuation of the GAO. Assessment of (the degree of) phosphate saturation of soil can be obtained by means of soil analysis according to the 'Protocol phosphate-saturated soils' (van der Zee et al. 1990). The above phosphate regulations are equally applicable to all materials covered by the decree on use of other organic fertilizers, and hence to the different types of compost.

Table 18.2. Application rates for animal manure, expressed as kg P_2O_5 ha/yr.

Period	Grassland	Silage Corn	Arable land
1-5-'87/1-1-'911	250	350	125
1-1-'91/1-1-'95	200	250	125
1-1-'95	175	175	125
2000	final rate	final rate	final rate

In all cases where soil is used as a waste management system the problem of P-saturation in relation to soil heterogeneity should be given special consideration. The heterogeneity aspects refer to the local variations in P bonding capacity of the soil (expressed as oxalate extractable iron and aluminum) and to the present degree of saturation. This heterogeneity, in combination with the large risk of phosphate leaching, makes it unappropriate that calculations are performed on the basis of average values. An extensive study in this area by van der Zee (1987) has shown how use can be made of stochastic approaches.

18.4 Potassium reactions in soil

The last major plant nutrient element, namely potassium, occurs in soil as a monovalent cation, and hence its behaviour is relatively easy to describe as compared to nitrogen and phosphorus. For equilibrium conditions, the relative occupation of the cation exchange complex of soil with K^+ can be arrived at by using standard equations of Kerr and Gapon. Cation exchange induces retardation of transport, which can be expressed by a retardation factor, to be derived from exchange isotherms (Bolt & Bruggenwert 1976).

Another situation arises when so-called fixation occurs, which may be found for K^+, and NH_4^+ as well, in clay minerals of the illitic type. Clay minerals with 'hexagonal' holes between the O-atoms (belonging to the 2:1 clay mineral types) show some preference for specific cations (K, NH_4, Rb, Cs) which in dehydrated form 'fitt well' in these holes. The result is a somewhat higher selectivity coefficient for these ions as compared to other monovalent cations, for example Na.

Illites belong to the mica types of minerals. They have the ability to 'lock in' considerable amounts of K-ions in the hexagonal holes, which may several times exceed the cation exchange capacity of the exterior sites of the clay particles. Intensivation of agricultural use of many alluvial soils, consisting of illitic type minerals, faces the problem of K-fixation resulting in too low availability of this nutrient for plant growth. Because of the competition for fixation between NH_4 and K, this may be remedied in part by simultaneous fertilization of potassium and nitrogen in the form of NH_4.

In case of complete fixation, the retardation of the ions in soil become infinity and hence the mobility zero. However, for other conditions the transport of K in soil can be described by the normally applied theory of translocation of solute compounds in porous, adsorbing media.

One of the problems with too high availability of K in pasture is the occurrence of grass tetany with cattle. Extra addition of Mg to the diet may remedy this.

As known so far, no negative results occur from high K levels in groundwater to be used for drinking water purposes. Nevertheless a K standard for groundwater has been introduced within the EC, at 40 mg K per liter. This may lead to problems with groundwater quality in the near future in certain areas where K leaching results in higher values. The above is an example of a situation where 'officially' an environmental problem will be introduced as the result of application of a standard without sufficient scientific background.

18.5 Limitations in waste management by occurrence of heavy metals

As mentioned before many waste products applied to soil show a severe imbalance between major plant nutrient elements and heavy metals. One of the most pronounced examples is pig slurry, which, although in itself not a waste, is sometimes brought onto soil in excessive amounts. In the past these slurries were rich in Cu because of copper addition to the pig feed in order to improve the feed conversion rate. An official EC regulation has lowered the copper addition to about one quarter of the original value, which has considerably improved the situation. Nevertheless the imbalance as mentioned still exists and thus large pig manure application will lead to enhanced damage to soil fertility. The same holds for waste products like sewage sludges and compost.

In the following, the attention will be focussed on the approach as followed in The Netherlands in the framework of the Soil Protection Act, introduced on January 1, 1987. Because this act is a skeleton law, implementation is achieved by G.A.O.'s and Decrees on use of specific materials on soil. From these, compost is taken here as an example. Although the sequel thus is limited to the Dutch case with respect to compost, the approach is of general validity for other situations and other waste materials.

First of all quality criteria of different types of compost with respect to heavy metal content have been set, as shown in Table 18.3, together with maximum application rates, presented in Table 18.4.

Table 18.3. Maximum values of some major heavy metal contents (mg/kg dry matter) in the different compost qualities as distinguished in the Dutch compost decree. A minimum organic matter content of 20% on dry matter basis is required for all composts.

Metal	Compost A[1]	B[2]	Compost quality Clean compost C[3]	Very clean compost
Cd	2	1	1	0.7
Cr	200	50	70	50
Cu	300	60	90	25
Hg	2	0.3	0.7	0.2
Ni	50	20	20	10
Pb	200	100	120	65
Zn	900	200	280	75
As	25	15	15	5

1) until 1-1-1995.
2) as from 1-1-1995.
3) until 1-1-1995.

Table 18.4. Maximum application rates (in metric tons of dry matter per hectare) of different types of compost for different soil uses.

Soil use	Compost (until 1-1-'95)	Clean compost (until 1-1-'95)	Compost (as from 1-1-'95)
grassland	1.5/y or 3/2 y or 6/4	3/y or 6/2 y	1.5/y or 3/2 y or 6/4 y
arable land and silage corn	3/y or 6/2 y or 12/4 y	6/y or 12/2 y	3/y or 6/2 y or 12/4 y
other soil use	6/y	12/y	6/y

These data may now be evaluated against the background of sustainable soil use.

One of the aims of soil protection regulations should be to ensure that agriculture is environmentally sustainable. This means that no adverse short- or long-term effects on soil, ground water, or crop quality may occur. Among others in the context of compost use, this implies constraints with regard to the balance of contaminants present in compost.

When a ploughed layer is considered the balance for a particular contaminant (e.g. heavy metal) reads:

$$\frac{dT}{dt} = \frac{dI}{dt} - \frac{dJ}{dt} - \frac{dP}{dt} \qquad (2)$$

where t is time in years, T is the accumulated amount in the ploughed layer, I is the annual input of contaminant, J is the annually leached quantity into groundwater, and P

is the annual plant uptake. Hence as well as the other terms have the dimension (mass/[area.depth.year]) where depth typically stands for ploughing layer thickness. When the input rate (dI/dt) exceeds the removal by leaching and plant uptake (and harvesting), accumulation in soil occurs. When the input and removal rates balance each other, no accumulation occurs and the content in soil (T) is in steady state. For a more or less constant input rate (due to atmospheric deposition and contamination of fertilizer with heavy metals) it usually takes a long time to raise T from negligible values to the level of soil quality standards (van der Zee et al. 1990).

The complication in evaluating (2) is that the accumulation (e.g. by adsorption), leaching rate, and plant uptake all depend on the concentration in solution, c. Hence, to resolve (2), the functions that express the relationships between T, I, J, P and c are needed. Whereas often the functions J(c) and P(c) may be linear (Boekhold & van der Zee 1991), the function T(c) is usually non-linear for heavy metals (Lexmond 1980; Christensen 1981; De Haan et al. 1987). This implies that except for special cases, (2) has to be calculated numerically. However, mostly the mentioned functions are inadequately known. Therefore it is usually impossible to calculate with sufficient accuracy whether a particular constant input rate leads first to exceeded crop quality, ground water quality, or soil quality standards, and at what time this will occur. For illustration, an example where such an assessment is feasible (for cadmium) is shown in Figure 18.6 (Boekhold & van der Zee 1991). Because this work is readily available details given in that reference are omitted here.
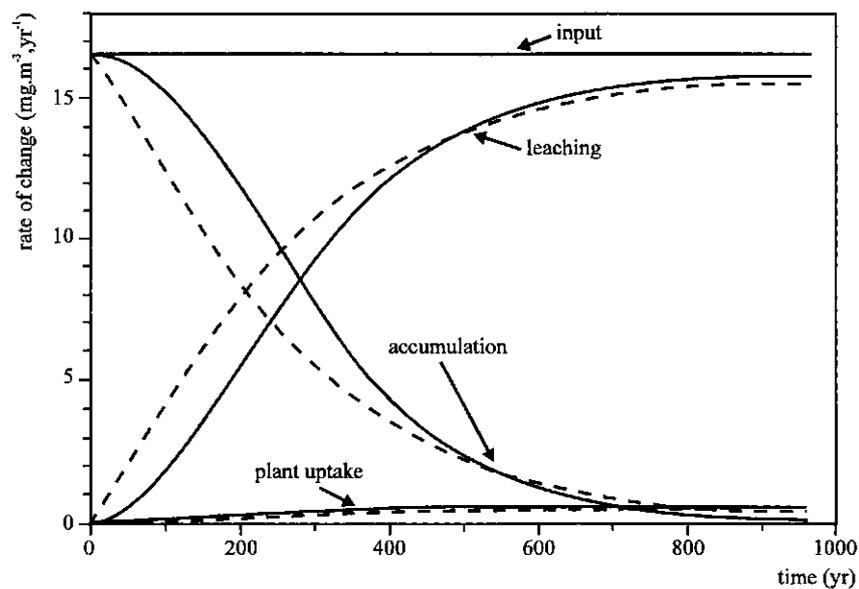

Figure 18.6. Cadmium balance as a function of time for a sandy soil and uptake by barley. Solid lines: analytical solution; dashed lines: numerical solution of the balance equation. From Boekhold & van der Zee 1991.

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As an alternative the definition of sustainability may be reconsidered. It implies that even at steady state the balance should be in agreement with existing crop, soil, and ground water quality standards. Hence, a particular currently accepted application rate of compost and fertilizer and diffuse atmospheric heavy metal deposition may not give rise to exceeded standards in the future. At least for crop and ground water quality this situation can be more easily evaluated, as will be shown later.

For illustration of this approach, first the acceptable removal is quantified. Because available Dutch ground water quality standards are maximum permissible levels, the choice was made to take the 'stand-still' (no increasing contamination) principle for ground water as the point of departure for acceptable leaching rates. Therefore, concentrations may not increase beyond mean values currently measured in ground water (Ferdinandus et al. 1989). Multiplication of these concentrations with the net precipitation (0.25 m^3/m^2 year) yields the acceptable leaching losses of Table 18.5. The dimension is (mg/m^2 year) because it was assumed that no accumulation occurs between the ploughed layer and 1 m depth (otherwise it should be (mg/m^3 year)).

Table 18.5. Calculated acceptable removal of heavy metals by leaching and by harvesting different crops.

Removal by	Acceptable removal in mg/m^2 year			
	Cd	Cu	Pb	Zn
Leaching	0.1	1.3	1.3	7.5
crop-harvest for				
(i) human consumption	0.03 - 0.25	3.6 - 11.0	0.1 - 0.5	36.0 - 110
(ii) animal fodder	0.15 - 0.75	3.6 - 13.5	7.0 - 31.0	36.0 - 135
vegetables*	0.43 - 0.85	3.7 - 7.5	1.7 - 4.5	37.0 - 75.0
Total (range)	0.13 - 0.95	4.9 - 14.8	1.4 - 32.3	43.5 - 143

* Relatively large values because of more harvests per year.

The acceptable removal in harvested plant material depends on the mean yield and acceptable tissue concentrations. Omitting details given elsewhere (Ferdinandus 1989), several vegetable rotation schemes and crops used for human consumption and for animal fodder were considered. Either quality standards (Pb, Cd) or the lowest tissue concentrations, where observed phyto-toxicity (Cu, Zn) was observed (MacNicol & Becket 1985), were used. Although, in view of differences in yields and acceptable concentrations, this gives only an average picture it serves for illustration. The results are given in Table 18.5. Observe, that for natural reserves the only removal is by leaching.

With regard to the input rates, it is necessary for an urbanized country like The Netherlands to take diffuse atmospheric deposition into consideration. Average data are provided in Table 18.6. To calculate the contribution of heavy metals due to compost and fertilizer applications, the fertilizer scenario needs to be known. With approximately 1.2% N, 0.6% P$_2$O$_5$, and 1% K$_2$O (percentage by mass dry matter) in vegetable, fruit and garden waste compost, it is obvious that the nutrient requirements can not be

Table 18.6. Heavy metal loads via atmospheric deposition, fertilizer and compost applications.

Source	Scenario*)	Mean load in mg/m² year			
		Cd	Cu	Pb	Zn
- Atm. Deposition	-	0.2	3.2	13.0	20.0
- Agriculture					
(i) Fertilizer	-	0.55	1.7	4.7	15.5
(ii) Compost + Fert.	1	0.59	10.0	15.9	36.4
	2	0.66	18.8	33	65
	3	0.88	36.6	62	124

*) see text.

met by compost application rates of 1.5-6 metric tons per year. Therefore it is assumed that the maximum permitted application rate of compost is combined with additional fertilizer applications. In Table 18.6 the resulting applications of heavy metals are given. In the calculations it was assumed that 1.5, 3 or 6 metric tons of compost were applied (scenarios 1, 2 and 3, respectively) per year per hectare. Furthermore, when this appeared necessary an additional fertilizer application was given to meet the demands for N, P_2O_5 and K_2O, assuming nitrogen was given as calcium ammonium nitrate which requires 0.46 kg CaO/kg N-fertilizer to neutralize its acidifying effect. The latter compound and the fertilizers also lead to heavy metal contamination as may be inferred from the case where no compost was applied. The crop requirements were assumed to be 140 kg N, 65 kg P_2O_5, and 84 kg K_2O (all per hectare per year) for agriculture. Usually, contamination by compost additions was the more important one. For vegetable-horticulture no calculations were provided in view of the wide variation in nutrients needed (2-4 times as much N, P_2O_5, and K_2O) and the large differences between crops in the ratios N: P: K. On average, the extra contribution of fertilizer to the heavy metal loads in horticulture equals 1.1 mg Cd/m² year and for Cu, Pb and Zn, 1.7, 6.5, and 30.0 mg/m² year, respectively, compared with the cases (i) and (ii1-3) in Table 18.6.

At this stage it is possible to compare the results of Tables 18.5 and 18.6. It is clear that the currently tolerated use of compost leads to positive balances for most scenarios

of Table 18.6. Consequently, it may be concluded that application rates of compost as indicated in Table 18.3, using a tolerated composition of compost as given in Table 18.1 column B, is not sustainable. Accumulation in soil, increased leaching rates, unacceptable product quality or a combination of these phenomena must be anticipated at the long term.

For a consistent set of environmental regulations of all compartments (atmosphere, soil, ground- and surface water) constraints should be set, $e.g.$ for compost use, on the contaminant that is the first to lead to problems. With regard to fertilizer use, the concept of 'discrepancy factor' may be introduced to facilitate the analysis which contaminant is most abundant (Scholtens & van der Zee 1991). The discrepancy factor F_d for essential elements (nutrients) may be defined as the ratio of application rate R_a over the removal rate R_r

$$F_d = \frac{R_a}{R_r} \tag{3}$$

where the removal rate should be taken equal to the uptake that is in agreement with a good product quality and the acceptable leaching. When F_d exceeds unity an excessive amount is applied. For a particular yield and optimal crop composition, F_d can be calculated. Because F_d for phosphorus applied with animal manure to soil exceeded the F_d-numbers for nitrogen and potassium, Dutch manure regulations were based on the phosphorus-balance of soil. As defined by (3), F_d exceeds unity (and is in fact infinitely large) for all non-essential elements, because the highest quality product would contain zero contaminants.

To make the F_d-concept more general and also useful for non-essential elements, R_r in (3) may be redefined as the uptake rate in agreement with acceptable product quality. Theoretically, the F_d-concept may be used to assess which of the considered heavy metals leads to environmental problems first.

As an illustration of the presented concept, R_r is calculated as the total rates given in Table 18.5 (using the lowest values of these ranges as those reflect standards for human consumption best). The application (or input) rate is obtained by summing the atmospheric deposition with those compost scenarios that may be acceptable ($i.e.$, Table 18.6 (ii)-1 and (ii)-2). The discrepancy factors are given in Table 18.7, and reveal that is F_d largest for Pb.

Table 18.7. Discrepancy factors F_d for two different cases of compost and fertilizer applications.

Scenario	Cd	Cu	Pb	Zn
1	6.0	2.7	20.6	1.3
2	6.6	4.5	32.9	2.0

This suggests that the greatest problems will be encountered for this element. It does not, though, imply that the problems for Pb are manifest earlier than for other heavy metals, because it is not clear whether for example leaching of Pb and the other metals

occurs at the same time. It does imply, that for the given scenarios the compost applications should be restricted based on Pb. If this is done, no problems are expected for the other elements. Approximately, the acceptable compost application rate of 1.5 metric tons/ha yr as given in Table 18.4 should be reduced by a factor twenty based on the discrepancy factor of Pb.

It needs to be mentioned that the given estimates should be applied after taking into consideration that:

1. even if no compost is applied, the required use of fertilizer may lead to F_d-factors exceeding unity, where another heavy metal may become limiting;
2. soil quality as such has not been considered, *i.e.*, adverse affects on the soil ecosystem were not accounted for;
3. depending on the relative importance of plant uptake and leaching, and the atmospheric deposition rate (which is difficult to influence), the F_d-factors for different elements may have to be significantly different to yield a conclusive strategy.

In conclusion of the above it may be stated that:

- Results of balance calculations for heavy metals, using average values, turn out to be discouraging for compost use.
- In part this may be due to the chosen sharp criterium: should production for human consumption be feasible everywhere?
- In part this is also due to diffuse (atmospheric) contamination. It seems obvious to reduce Pb-emissions. (Currently the standard of 200 mg/kg is not met). Atmospheric emissions are apparently not taken into account in the compost regulations.
- Spatial variability of diverse contamination fluxes was not taken into account: in urban areas Pb-fluxes may significantly exceed the average, in rural areas the opposite may be true. Nevertheless, the calculations provide a warning that current compost regulations may even violate the concept of sustainability on average.
- In particular care should be taken with regard to allotment-gardens because:
  . very clean compost is not that much different in quality;
  . the application rates may be large;
  . due to their location in urban areas, the atmospheric deposition may be relatively large;
  . exposure to poor quality crops is systematically large due to consistent consumption by the same producers.

The general conclusion when considering soil as a waste management system, unfortunately, must be that, although many wastes do contain valuable constituents, their use is severely restricted from a soil protection point of view.

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18.6 References

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