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ORIGINAL PAPER

Effects of soil drying and rate of re-wetting on concentrations and forms of phosphorus in leachate

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Abstract The drying and re-wetting of soils can result in the modification of the amounts and forms of nutrients which can transfer, via leachate, from the soil to surface waters. We tested, under laboratory conditions, the hypothesis that the rate of re-wetting of a dried soil affects the solubilisation and concentrations of different forms of phosphorus (P) in leachate. A portion of grassland pelostagnogley soil (sieved moist <2 mm) was dried at 35°C and another portion maintained at approximately 40% water-holding capacity. Water (25 ml) was added at ten regularly spaced time intervals in 2.5-ml aliquots to the surfaces of both soils over periods of 0, 2, 4, 24 and 48 h, resulting in different rates of application. The leachate was collected and analysed for dissolved (<0.45 μm) and particulate total P and molybdate reactive and unreactive P. The rate of re-wetting significantly changed the concentrations of P, especially dissolved forms, in the leachate. Dissolved P concentrations were highest in leachate from the 2-h treatment, while particulate P concentrations were highest in the 0-h treatment leachate. In all cases, most P was unreactive and, therefore, likely to be in an organic form. Soil drying decreased microbial biomass, but this could not be directly linked to an increase of P in leachate. These results suggest that changes in patterns of rainfall frequency and intensity predicted by climate change scenarios could significantly affect the quantities of P leached from soils.

Keywords Drying–re-wetting · Phosphorus · Leachate · Soil microbial biomass

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Introduction

Drying and re-wetting is one of the most common and widespread forms of abiotic stresses experienced by soils (Soulides and Allison 1961). Effects on the rate of microbial turnover have long been known, with Birch (1958) being widely attributed as the first to have reported the pulse of carbon dioxide emitted from dried soils following re-wetting, indicating an increase in the rate of microbial turnover. Others have reported similar findings (Bottner 1985; Fierer and Schimel 2002, 2003; Mikha et al. 2005), but the actual processes involved have been subject to debate. Originally, it was thought that physical soil processes could explain these flushes of activity. For example, desiccation of soil organic matter results in exposure to enzymes of previously inaccessible surfaces of organic or organo-mineral colloids due to aggregate fragmentation or increased porosity (Birch and Friend 1958; Soulides and Allison 1961). Powlson and Jenkinson (1976) confirmed that the solubilization of organic compounds was caused by physical disruption of the soil structure and substrate desorption from surfaces, in addition to increased microbial mobility and diffusion of soluble organic compounds, all caused by drying–re-wetting cycles. More recently, attention has focused on the release and

cycling of nutrients from the soil microbial biomass (biomass) following drying–re-wetting, despite it being a relatively small fraction of total organic matter in most soils (McNeill et al. 1998; Turner and Haygarth 2001). Due to the potential of phosphorus (P) to trigger eutrophication events, there is increasing interest in whether soil microorganisms are an important source of predominantly dissolved P which may transfer in significant quantities to surface waters. However, the factors affecting solubilization and transfer of P from soil microorganisms are poorly understood. Turner and Haygarth (2001) showed a relationship between the quantity of P mobilised in re-wetted air-dried soils and P in the biomass, and suggested this may help explain the solubilisation of P in soils that may be subsequently transferred to watercourses. McNeill et al. (1998) and Turner et al. (2003a) showed that bacterial cell lysis was an important source of mobilised P in Australian grassland soils and a potentially important source of plant-available P. Turner et al. (2003b) postulated that organic P in streams in upland northern England originated from the microbial biomass, being released in pulses following drying–re-wetting and freeze–thaw cycles, and is crucial to the maintenance of the trophic status of upland streams. However, there is currently no conclusive evidence that there is a direct link between P release from the microbial biomass and P in surface waters.

Factors such as degree (West et al. 1992; Kieft et al. 1987), rate (Chao and Alexander 1984; Roberson and Firestone 1992), duration (De Nobili et al. 2006) and frequency (Bottner 1985) of drying affect soil microbial responses and, therefore, potential solubilisation of P. However, the effect of the rate at which a dried soil is re-wetted is, as yet, unreported. We compared the concentrations and forms of P released in leachate following the addition of similar quantities of water at different rates to both dried and moist soils. This experiment tested the hypothesis that the rate of re-wetting of a dried soil influences the solubilisation and leaching of P from the soil microbial biomass. The potential implications are that predicted changes in patterns of precipitation due to climate change could result in changes in the concentration and source of P released from soils following natural drying–re-wetting cycles (although there are also implications for irrigated soils), and consequent effects on surface water quality.

Materials and methods

Soil type, collection and preparation

Soil was collected in October 2007 from North Wyke Research Station, Southwest England, UK. This soil is a clayey, non-calcareous typic haplaquept (USDA) of the Hallsworth Series (FAO dystric gleysol), overlying clay shales of the Crackington Formation (Harrod and Hogan 2008). The main properties of the soil were total carbon (C), organic C, nitrogen (N) and P contents of 4.0%, 3.7%, 0.5% and 0.1%, respectively. It had a clay content of 38% and a pH of 5.3 (Harrod and Hogan 2008). Approximately 1 kg of soil (0–10 cm depth) was collected and stored at 4°C until used. The soil was prepared as described by Vance et al. (1987) for the measurement of soil microbial biomass C. In summary, soil was crumbled and all visible non-soil material (e.g. stones, roots, leaves, earthworms, etc.) was removed. The soil was sieved (<2 mm) and then divided into two equal size sub-samples, one of which was dried at 35°C to a constant weight and the other held at field moisture content. Both soils were stored at 4°C for 1 week until use.

Rate of re-wetting

The dried and moist soils were pre-incubated at 25°C for 24 h prior to use. Three sub-samples of the moist and dried soil were dried to a constant weight at 100°C to determine soil moisture content. Subsequently, three replicates comprising 21 g of the dried soil were placed into 50-ml plastic, conical funnels with a maximum internal diameter of 63 mm, a spout of 11 mm diameter and sides sloping at an angle of 60° to the horizontal. Each spout was plugged with 0.3 g of glass wool. Each replicate was irrigated with deionised water in 5-ml aliquots until enough water had been added to generate a minimum of 10 ml of leachate, the amount required to enable measurement of total P (TP) and molybdate reactive P (MRP) in filtered and unfiltered leachate samples. Twenty five millilitres of water was required for this. Subsequently, 15 replicates each of the dried and moist soils were prepared comprising 21 g dry weight equivalent (DWE) of soil placed into 50-ml funnels plugged with glass wool. The soil was loosely packed by gently tapping the funnels. Whatman 0.45-μm cellulose nitrate membrane filter papers (Whatman, Clifton, NJ, USA) were placed on the soil surfaces to facilitate even distribution of irrigation water. Three randomly selected replicates of dried and moist soils were irrigated with a total of 25 ml of water added in 2.5-ml aliquots spaced evenly (total of 10 applications) over total incubation periods of 0, 2, 4, 24 and 48 h. Hence, for the 2-h re-wetting rate, 2.5 ml of water was added every 12 min; for the 4-h re-wetting rate, an application was made every 24 min, and so on. In the case of the 0-h treatment, 25 ml of water was added slowly but in one application. In all cases, water was applied to the filter papers on the soil surfaces by carefully expelling the required aliquot of deionised water from 25-ml syringes. Leachate was collected in 30-ml polycar-

bonate vials and stored at 4°C for no longer than 24 h, before half the leachate was filtered through Whatman 0.45 μm cellulose nitrate membrane filter papers and analysed for MRP as described below. The remaining leachate was stored at 4°C prior to analysis for TP. All incubations were carried out at a constant temperature of 18°C. Following the final irrigation, replicates were left for 15 min to enable leachate collection and then were analysed for microbial biomass carbon (Vance et al. 1987). The original method was modified to deal with smaller soil weights than were used by Vance et al. (1987), but ratios of all components were maintained. Snars et al. (2006) demonstrated with the same soil used here that, if ratios of components remain the same, accurate measurements can still be made using different quantities of soil. The soil from each replicate was weighed and divided into seven equal sub-samples. Three sub-samples were extracted immediately in 12 ml 0.5 M K_2SO_4, and three were fumigated for 24 h with chloroform prior to extraction. For each treatment, the remaining sub-samples were oven-dried to determine soil moisture contents. Biomass C was calculated according to Vance et al. (1987).

Laboratory analyses

Bicarbonate extractable MRP and water extractable MRP were measured on both dried and moist soils prior to incubation. For bicarbonate extractable MRP, 10 g DWE of soil was extracted in 200 ml of 0.5 M $NaHCO_3$ (adjusted to pH 8.5 with NaOH) on a reciprocating shaker for 1 h then filtered through pleated Whatman No. 42 filter papers (Whatman). For water-extractable MRP, 10 g DWE of soil was extracted with 40 ml of deionised water on a reciprocating shaker for 1 h, and then filtered as above. Half of each leachate was filtered through a Whatman 0.45 μm cellulose nitrate membrane filter (Whatman). Filtered and unfiltered leachate samples were digested (Rowland and Haygarth 1997) and was TP determined. In all cases, orthophosphate was measured according to Murphy and Riley (1962) using a BioTek Synergy HT PR101 plate reader (BioTek, Bad Friedrichshall, Germany). MRP in filtered and unfiltered leachate samples was measured similarly, but in non-digested samples.

Unreactive P was calculated as the difference between total and MRP in the filtered and unfiltered samples, respectively. Unreactive P is generally considered to be organic P (Haygarth et al. 1998). The filtered fraction is considered to be dissolved P (Haygarth et al. 1997), and the difference between concentrations in the filtered and unfiltered samples is termed the particulate fraction. Consequently, data are presented as particulate and dissolved total, unreactive and reactive P. Additionally, the total amount of P in unfiltered leachate was measured (TP in unfiltered sample). The amount of C in the K_2SO_4 extracts was determined as described by Vance et al. (1987). Total soil C and N were measured simultaneously using an elemental analyzer (Carlo-Erba NA2000, Milan, Italy). Total soil P was measured using an Accuris inductively coupled plasma optical emission spectrometer (ARL/Fisons, Eclubens, Switzerland) after aqua regia acid digestion. Soil pH was measured using a Jenway 3320 pH meter according to the Ministry for Agriculture, Fisheries and Food (1986). Soil water potentials of both the dried and moist samples were measured using a Decagon WP4-T Dewpoint Potentiameter.

Data analyses and statistics

The significance of differences between treatments was assessed by one-way analysis of variance using Genstat v. 10.1 (VSN International, Hemel Hempstead, UK). Significant differences are reported at the $p<0.05$ level. Significant differences between individual mean values were calculated using the variability between replicates using an adjusted two-sample t test (Cochran and Cox 1950). The data in all tables and text are given as the mean ± standard error unless indicated otherwise.

Results

Initial water extractable MRP concentrations from both the dried and moist soils were similar: 204±7 and 204±9 μg P kg^{-1}, respectively (Table 1). The bicarbonate extractable reactive P concentration in the dried soil (21.3±0.2 mg P kg^{-1}) was almost double that in the moist soil (12.4±0.5 mg P kg^{-1}) (Table 1). The initial moisture contents of the dried and moist soils were 0.9±0.2% and 24.2±0.0%, respectively, resulting in soil water potentials of −78 and −0.4 MPa, respectively (Table 1).

Phosphorus concentrations in the deionised water used for irrigation were generally low, but the filtering process increased them. Unfiltered blank leachate contained 16.2±3.3 μg TP L^{-1} and 1.7±0.6 μg MRP L^{-1}, and filtered blank leachate contained 36.3±7.2 μg TP L^{-1} and 4.2±0.0 μg MRP L^{-1}.

Table 1 Initial MRP and moisture properties (±standard error) of moist and dried soils

	Moist soil	Dried soil
Water extractable MRP, mg P kg^{-1}	204±7	204±9
Bicarbonate extractable MRP, mg P kg^{-1}	12.4±0.5	21.3±0.2
Moisture content, %	24.2±0.0	0.9±0.2
Soil water potential, MPa	−0.4	−78

Microbial biomass C concentrations in the dried soils ranged from 179±113 mg kg^{-1} in the initial sample to 502±92 mg kg^{-1} in the 2-h soil, while in the moist soils, it ranged from 504±75 mg kg^{-1} in the 2-h sample to 682±24 mg kg^{-1} in the 0-h sample (Fig. 1). The rate of irrigation caused no significant differences in microbial biomass C in either the dried or moist soils. At all irrigation rates, microbial biomass C in dried soils was always significantly lower than in the equivalent moist soils, with the exception of the 2-h soils, where the microbial biomass C concentrations were similar (moist soil=503±75 mg kg^{-1}, dried soil=502±92 mg kg^{-1}).

TP concentrations in unfiltered leachates from the dried soils were significantly greater than in those from the moist soil at all irrigation rates (Fig. 2). TP concentrations in the dried soil leachates ranged from 1,254±24 μg P l^{-1} in the 0-h sample, gradually declining to 581±133 μg P l^{-1} in the 48-h sample. TP concentrations in the moist soil leachates ranged from 372±157 μg P l^{-1} in the 4-h leachate to 109±35 μg P l^{-1} in the 48-h leachate. Changes in irrigation rate caused no significant differences in TP concentrations in unfiltered leachate from the moist soils, but in the dried soils, some significant differences did occur. Most notably, the 0-h leachate TP concentration was significantly higher than that in the 2-, 4- and 48-h leachates.

Overall, there were no significant differences between P concentrations of all particulate P fractions in leachate from the dried and moist soils (Fig. 3a–c). However, all particulate P fraction concentrations in leachate from the moist samples were similar, with maximum concentrations in the 4-h leachates and minimums in the 48-h leachates. Also, all particulate P fraction concentrations in the 2- and 4-h leachates from the moist soils were greater than those

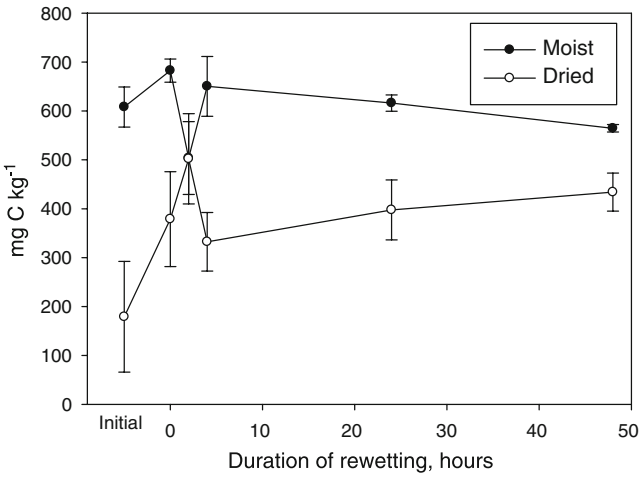

Fig. 1 Microbial biomass carbon (*C*) after re-wetting soil at different rates. *Error bars* (±standard error) calculated from three experimental replicates, except for initial samples, which represent errors from analytical replicates

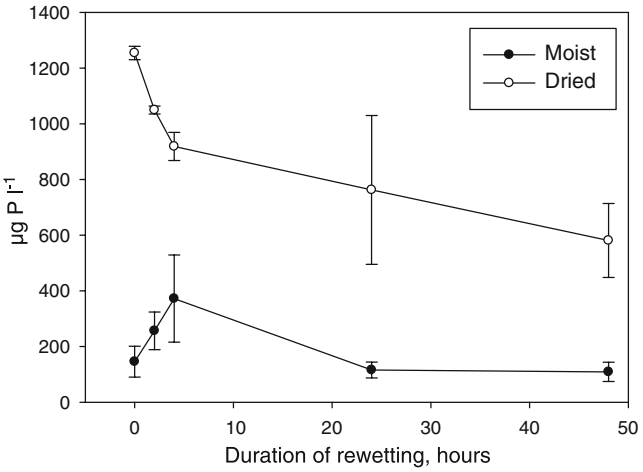

Fig. 2 Total phosphorus (*P*) in unfiltered leachate samples. *Error bars* (±standard error) calculated from three experimental replicates

from dried soils, whilst in the 24- and 48-h leachates, the opposite occurred. These differences were significant only for MRP in the 2-, 4- and 24-h samples. In the dried soil leachates, the particulate TP concentration (Fig. 3a) was highest in the 0-h leachate (536±81 μg P l^{-1}) and lowest in the 2-h leachate (111±94 μg P l^{-1}), whilst in the moist soil, the maximum and minimum concentrations were in the 4-h (376±158 μg P l^{-1}) and 48-h leachates (85±23 μg P l^{-1}), respectively.

The particulate unreactive P concentrations in leachates (Fig. 3b) followed the same pattern as the particulate TP concentrations. Maximum concentrations in leachate from the moist and dried soils were 293±161 μg P l^{-1} in the 4-h leachate and 499±85 μg P l^{-1} in the 0-h leachate, respectively, while minimum concentrations were 65±24 μg P l^{-1} in the 48-h sample and 95±90 μg P l^{-1} in the 2-h sample, respectively.

Particulate MRP concentrations in leachate (Fig. 3c) were generally an order of magnitude lower than the equivalent total and unreactive P concentrations. In the moist soil leachates, P concentrations initially increased to a maximum in the 4-h leachate (82±3 μg P l^{-1}) and then declined to a minimum in the 48-h leachate (19±7 μg P l^{-1}). In the leachate from the dried soil, almost the opposite occurred, with the maximum particulate MRP concentration occurring in the 48-h leachate (45±6 μg P l^{-1}) and the minimum in the 4-h leachate (13±7 μg P l^{-1}).

The dissolved P fractions in leachate demonstrated different results to the particulate P fractions. In all cases, the dissolved unreactive and TP in leachate from the dried soils contained significantly more P than that from the corresponding moist samples (Fig. 4a–c). For each P fraction from the dried soil, the maximum concentration occurred in the 2-h leachate. Temporal patterns in P concentrations in leachate from the moist soils were more

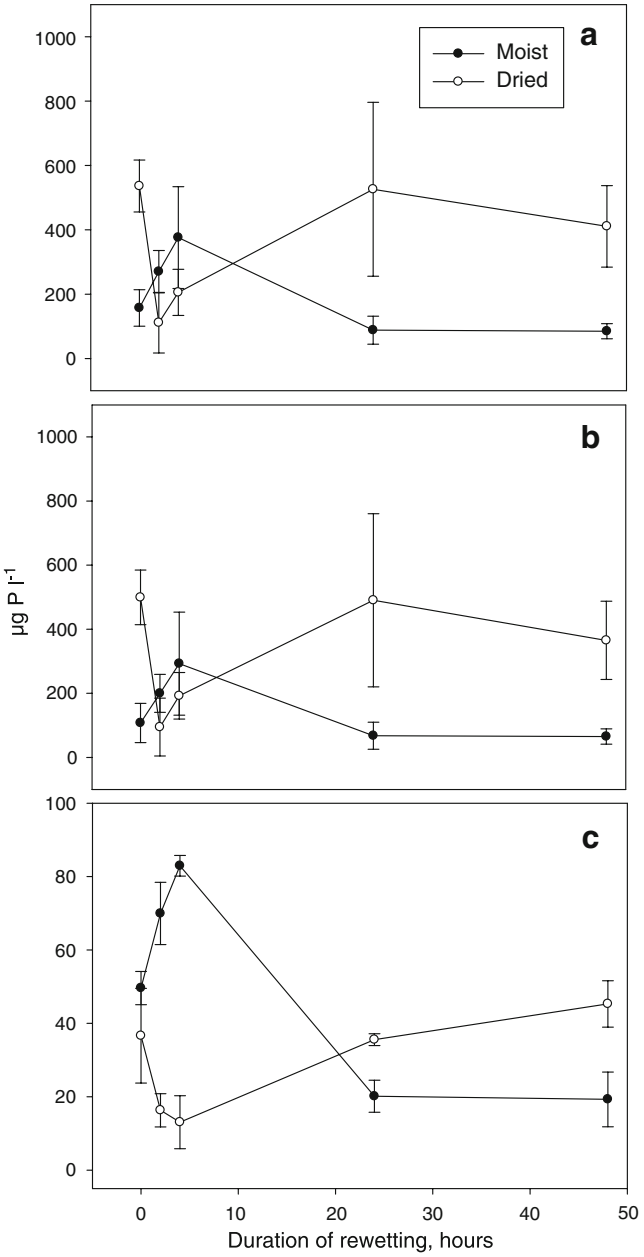

Fig. 3 **a**–**c** Particulate phosphorus (*P*) fractions (**a** TP, **b** unreactive P, **c** MRP) in leachate. *Error bars* (±standard error) calculated from three experimental replicates

variable, and in some cases, the more rapid re-wetting rates resulted in negative concentrations of dissolved total and unreactive P, probably reflecting some adsorption of the small quantity of P present in the deionised water used to irrigate the soils. Dissolved TP concentrations (Fig. 4a) in leachate from the dried soil were highest in the 2-h sample (938±90 µg P l^{-1}) and lowest in the 48-h sample (170±13 µg P l^{-1}), while in the moist soil leachate samples, maximum and minimum concentrations were in the 24-h sample (28±41 µg P l^{-1}) and 2-h sample (−13±2 µg P l^{-1}), respectively.

The dissolved unreactive P (Fig. 4b) and dissolved TP (Fig. 4a) concentrations in leachates followed similar trends. Maximum unreactive P concentrations in leachate from the moist and dried soils were 26±42 µg P l^{-1} in the 24-h leachate and 900±87 µg P l^{-1} in the 2-h leachate, respectively, while minimum concentrations were −23± 1 µg P l^{-1} in the 2-h leachate and 169±13 µg P l^{-1} in the 48-h leachate, respectively.

Dissolved MRP concentrations in leachate (Fig. 4c) were generally an order of magnitude lower than the equivalent total and unreactive P concentrations, as were the particu-

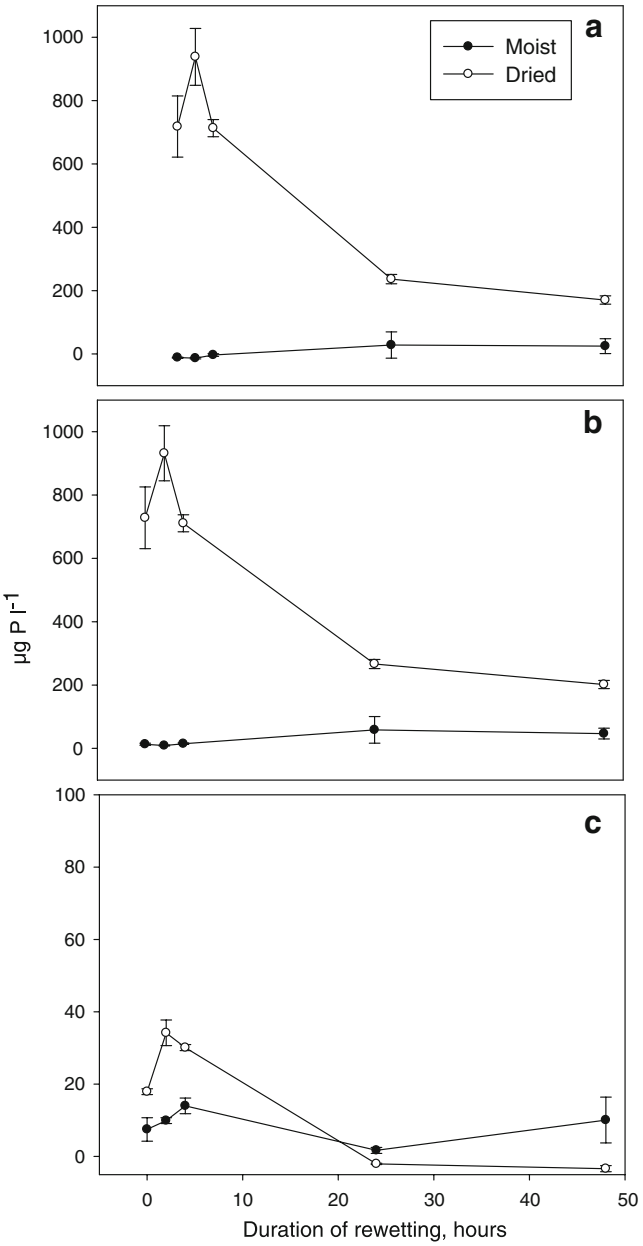

Fig. 4 **a**–**c** Dissolved phosphorus (*P*) fractions (**a** TP, **b** unreactive P, **c** MRP) in leachate. *Error bars* (±standard error) calculated from three experimental replicates

late P fractions. In the moist soils, MRP concentrations initially increased to a maximum in the 4-h leachate (14.0± 2.1 μg P l^{-1}) and then declined to a minimum in the 24-h leachate (1.7±0.8 μg P l^{-1}). In the leachate from the dried soil, the maximum MRP concentration was in the 2-h leachate (34.2±3.5 μg P l^{-1}) and the minimum in the 48-h leachate (−3.4±0.8 μg P l^{-1}).

Discussion

The results reflect complex interrelationships among chemical, physical, and biological soil properties. The drying was relatively extreme in terms of air-drying, but 35°C is reported to be representative of high temperatures experienced by soil surfaces during summer in the UK (Wild 1988). Although drying at this temperature increased the bicarbonate extractable MRP in the soil, it did not increase the quantity of water extractable MRP, suggesting that most of the additional bicarbonate extractable MRP was sorbed by the soil, so it was not leached. Hence, in this soil, the drying process may change the availability of P to plants, but it does not appear to directly affect the availability of leachable MRP. However, drying increased dissolved MRP in leachate in the rapid re-wetting rates (0, 2 and 4 h, Fig. 4b), suggesting that it is not availability that affects MRP lost in leachate, but that other confounding factors affecting dissolved MRP transfer (as opposed to solubilisation) are more important, as discussed below.

The smaller microbial biomass in the initial dried sample compared with the initial moist sample (Fig. 1) indicates that drying is partially biocidal (Bottner 1985; Kieft et al. 1987; Van Gestel et al. 1993). However, the lack of microbial biomass recovery in the dried soils with slower re-wetting rates suggests that, at the rates investigated here, the re-wetting rate does not significantly affect remaining microbial biomass survival or recovery in the dried samples. Despite the fact that, generally, most of the P in the leachate from the dried soils at all re-wetting rates was unreactive, the microbial biomass changes do not support the hypothesis that it is a major source of the unreactive P because they do not demonstrate similar trends in concentration. In the dried soil, the overall effect on the microbial biomass of re-wetting, at the individual microbe level, is probably similar for all rates of re-wetting with regard to P release; changes only happen over different time frames, hence the lack of significant differences between microbial biomass in dried soils following re-wetting at different rates. This is also likely to be the case for other sources of P mobilised in the soil. However, the slower the re-wetting process, the more opportunity there is for recycling of any solubilised P before it is transported from the soil, meaning that the surviving microbial biomass, to some extent, could be regulating the P concentrations in leachate. Thus, changes in, for example, osmotic regulation and assimilation of mobilised P compounds by the surviving microorganisms under longer re-wetting rates could contribute to the observed trends. During drying, some organisms release solutes to protect against desiccation, which are a likely source of unreactive P (Halverson et al. 2000), and it is possible that, during re-wetting, the C:P ratio of the surviving microorganisms narrowed following re-uptake of P, which might have originated from the biomass itself and reduced the quantity of P available for leaching (He et al. 1997). Only a short period of time is required for uptake of P to commence upon re-wetting. Bushby and Marshall (1976) report the repair of bacterial cells damaged by desiccation occurring within an hour of rehydration, while De Nobili et al. (2006) report 'appreciable' synthesis of ATP only 6 h after the re-wetting of dried soils. Consequently, uptake of mobilised P by the surviving microorganisms may partially explain the trends observed. However, further studies are required to identify the actual source of the P in leachate. This could possibly be done using isotopically labelled bacteria, or by immediate measurement of microbial biomass P in the re-wetted soils. However, both of these methods have their limitations.

The impact of drying and rate of re-wetting on other pools of unreactive P in the soil must also be considered, and it is likely that these are at least as equally important as the biomass (Jenkinson 1966; Van Gestel et al. 1991; Pulleman and Tietema 1999). Physical stresses caused by drying can disrupt organic matter coatings found on soil particle surfaces (Bartlett and James 1980). Turner et al. (2002) found similar proportions of both non-biomass myo-inositol hexakiphosphate and microbially derived phosphate diesters in water extracts of air-dried pasture soils. The microbial community composition is also important, as different micro-organisms respond differently to patterns of drying–re-wetting, affecting the quantities of P released directly from microbes (Bushby and Marshall 1976; Fierer et al 2003). Additionally, microbial community composition can affect the form of microbially synthesised P in soil (Bünemann et al. 2008), which will also be affected differently by patterns of drying–re-wetting, thus influencing the quantity and form of P potentially lost in leachate.

The gradual decrease in TP concentrations in unfiltered leachates from the dried soils with slower re-wetting rates (Fig. 2) reflects the combined effects of changing concentrations of both dissolved and particulate P in leachate. The significance of soil drying and rate of re-wetting on particulate P concentrations in leachate are inconclusive, with no consistent effect on any fractions observed. Although drying decreases soil aggregate stability (Powlson and Jenkinson 1976; Utomo and Dexter 1982; Denef et al. 2001) and may promote particulate detachment and

transport in surface runoff (Sharpley et al. 1996; Haygarth and Jarvis 1999), this was not a factor in this experiment. Here, drying has not had a significant effect on the mobilisation and transfer of particulate P in leachate. However, this is not to say that physical processes are not important. At least some of the P found in leachate (both reactive and unreactive) is probably associated with the physical detachment and mobilisation of soil colloids within the soil (as opposed to only on the soil surface) by the drying–re-wetting process. These can contribute to both operationally defined dissolved (<0.45 μm) and particulate (>0.45 μm) P fractions because colloids range in size from 1 nm to 1 μm (Heathwaite et al. 2005). Additionally, while P mobilised from lysed microbial cells will be leached rapidly from the soil during the rapid re-wetting treatments, the slower re-wetting will allow some of this cell-derived P to be hydrolyzed by phosphatase and either sorbed to soil, taken up by microbes or leached. This could also be a reason for the decline observed in dissolved unreactive P concentrations with the slower re-wetting rates. It is also possible that the patterns seen in the particulate and dissolved P concentrations reflect the fact that some of the particulate unreactive P is in cell fragments which can survive long enough to be leached in both the rapidly and slowly re-wetted treatments, while the dissolved unreactive P would be more susceptible to hydrolysis during the slower re-wetting treatments (Hannapel et al. 1964).

There are few studies on the loss of P in leachate as a result of drying and re-wetting soil (Qiu et al. 2004). Most report the effects of drying on P concentrations in water extracts or soil solution (Turner and Haygarth 2001; Turner et al. 2003b; Venterink et al. 2004), rather than in leachate. Aggregate stability and pathways are not considered in the above experiments, which simply consider the solubilisation effects of drying and re-wetting, and not the mobilisation and transference factors. In our work, the sieving to <2 mm may have decreased the effect of aggregate stability, with the main physical impacts being on colloids and organic material resulting in mobilisation of particles of less than 0.45 μm diameter, and therefore classified as dissolved (Haygarth et al. 1997). However, drying and rate of re-wetting do have a significant effect on the concentrations of all dissolved P fractions. This is particularly important because dissolved P is the most likely form to be lost in leachate. With dried soil, the maximum concentrations of all dissolved P fractions were consistently measured in the 2-h samples, suggesting that the re-wetting rate is important. In the 24- and 48-h re-wetted soils, dissolved MRP in leachate was greater from the moist than from the dried soil. The MRP concentrations from the dried soils at these slower re-wetting rates were negative, representing complete removal from soil solution of any mobilised dissolved MRP, as well as the small quantities in the deionised irrigation water. Clearly, for MRP to be leached from the soil, it must be transported rapidly; otherwise, it will be fixed by the soil or immobilised by the microbial biomass.

Another factor affecting concentrations of particulate and dissolved P fractions could be differences in soil physical properties brought about by the different moisture contents at the start of the experiment, as well as temporal changes caused by the different irrigation rates. In the dried soil, the applied water would have diffused more evenly through the soil than in the moist soils due to the tightly held pore water in many of the interstitial spaces in the latter. Consequently, the water added to the dried soils would potentially contact more soil surfaces than the water added to the moist soils. Therefore, it would have had the potential to solubilise and transport more P from the soil in leachate. Also, the physical response of a dried soil to re-wetting is not necessarily immediate. Therefore, despite the same quantity of water being added to each sample, the final applications to the dried soils at each re-wetting rate would have been made to soils in different physical states (even in aggregates <2 mm), thereby affecting the pathways, rate of water flow and hence kinetic energy of the water, and consequently, the different concentrations and forms of P leached.

Conclusions

The rate of re-wetting a dried soil under laboratory conditions affects the concentrations and forms of P in leachate. Most of the leached P is dissolved and unreactive, with faster re-wetting rates giving rise to higher concentrations in leachate. It is likely that some of this P is derived from the microbial biomass, but other non-biomass sources are almost certainly also important. Soil drying decreases microbial biomass, but this could not be directly linked to the increases of P in leachate. These results suggest that predicted changes in patterns of rainfall as a result of climate change and, consequently, period of time taken for leachate flow to be stimulated from previously dried soils could affect the quantities of P leached from soils.

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