

**THE BIOTURBATION TRANSPORT OF CHEMICALS
IN SURFACE SOILS**

A Thesis

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ABSTRACT

Large quantities of chemicals, such as pesticides, fertilizers, and industrial wastes, have been found throughout the environment raising concerns due to their ecological impacts and implications to human health. Soil is the most important repository of many organic chemicals in the environment. The objective of the present study was to determine the importance of the soil solid phase in the transport of chemicals in soils, and yield quantitative information to better describe bioturbation and its role in the movement of soil particles.

Vertical movement of chemicals in the soil solid phase occurs by mixing mechanisms, such as bioturbation, cryoturbation, and dryoturbation. An extensive variety of soil-dwelling animals are responsible for bioturbation. Data on this process were located in the published literature, cataloged, and evaluated to estimate sorbed phase diffusion coefficients from soil turnover rates and effective depths reached by selected organisms. The impact of animals in soil processes varies depending on species, numbers, diversity, size, and feeding and burrowing behavior, which at the same time depend on soil properties, climate conditions, among others. Based on a 50% probability of occurrence, the approximated average depth of soil bioturbation was 20 cm. The periodic mixing of soil due to agricultural practices influences the sorbed phase transport of chemicals; representative numerical values of this type of “bioturbation” were estimated as well. Soil concentration profiles for selected PCBs were collected from literature and modeled. Model extracted sorbed phase diffusion coefficients of $4.03\text{E-}07$, $5.98\text{E-}07$, and $5.81\text{E-}07$ m^2/day were obtained for PCB-52, 153, and 101, respectively. These numerical chemical values were in agreement with bioturbation particle turnover values. For all congeners, percentage contribution of transport in the solid phase corresponded to more than 90% of the overall transport process.

The model exercise provided valuable insights into the relative importance of the different soil transport mechanisms. It was concluded that PCBs are transported principally in association with the soil solid phase; their transport in air and water phases is insignificant. Therefore, chemical fate and transport models must account for the mixing of soil particles by bioturbation as it greatly influences the transport of chemicals sorbed to them.

1. INTRODUCTION

Humans have produced numerous chemicals, and will continue to produce and use them all over the world. Likewise, tons of agricultural, commercial, industrial, and domestic wastes are daily produced. These chemicals and pollutants reach the natural environment through several pathways, having serious implications to both human health and biological receptors. Therefore, it is important to be able to trace their transport pathways and concentrations levels in the environment.

The understanding of how chemicals enter the environment, their fate and transport, and their environmental impact is essential to develop appropriate pollution prevention strategies. Chemicals released to the environment move across environmental boundaries and are therefore found in most media, including water, air, soil, groundwater, and vegetation.

Concentration of pollutants or chemicals can be obtained by either chemical fate and transport (CFaT) modeling or field measurements. This task demands an understanding of the physical, chemical, and biological processes that govern the movement of chemicals among the different environmental compartments (Cohen, 1998).

Soil is one of the most important and complex compartments, being considered as the major repository of many organic chemicals in the terrestrial environment (Mclachlan et al., 2002). Since it is not an isolated medium, its constant dynamic interaction with the larger environment facilitates the exchange of chemicals with water, air, groundwater, biota, and vegetation. This is part of the inherent multimedia nature of environmental pollution.

Figure 1 shows some of the pathways that connect soil with its surroundings, through which transport of substances can take place.

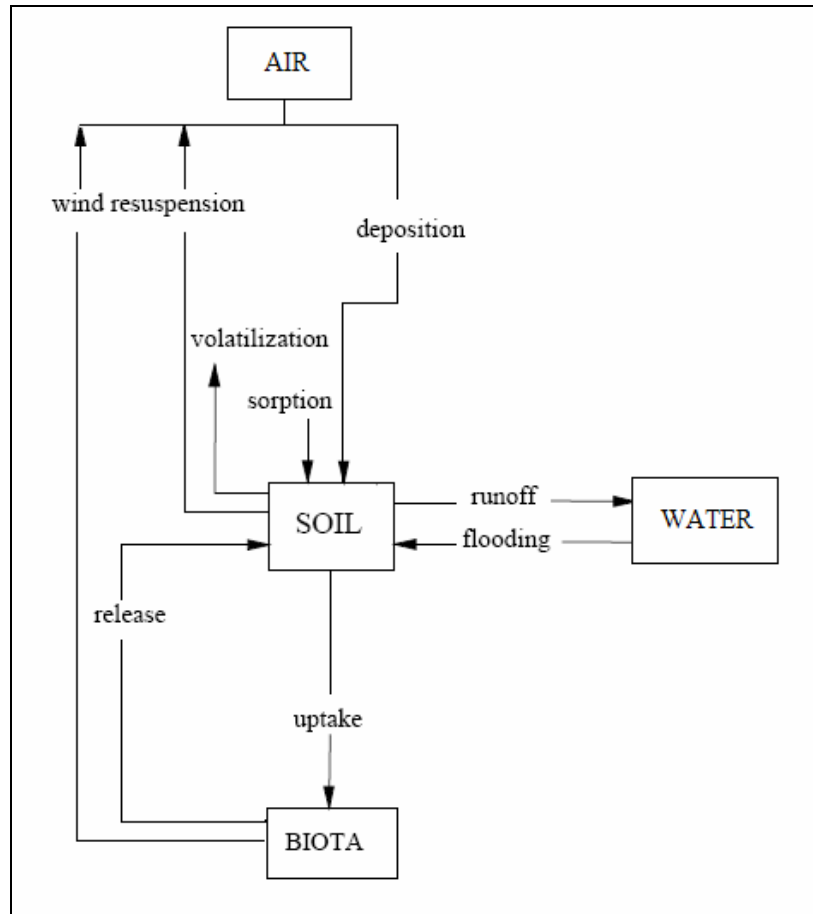


Figure 1. Soil Transport Pathways (Adapted from Cohen, 1998)

Humans are exposed to soil contaminants through their interaction with the multimedia environment showed in Figure 1. There is an uptake of soil contaminants by biota feeding on plants and on materials of plant origin, and consequently the same uptake through human ingestion of animal products, such as milk or meat. Also, there is direct exposure of humans to contaminated deposited materials in soils, and to contaminants released to the air.

The most conspicuous part of any soil is the surface zone. Through it, matter and energy are transported between the soil and the atmosphere. However, its surface does not necessarily represent the character of the whole soil (Hillel, 2004). To describe soil in detail, a soil profile must be examined.

The soil profile consists of a succession of distinct strata called horizons. The O-horizon has high percentage of organic matter, mainly layers of decaying plant (e.g., leaf matter) and animal tissue. The A-horizon or topsoil is the zone of major biological activity formed by mineral material generally enriched with organic matter. Underneath the A horizon is the B horizon, where some materials leached from the A horizon are accumulated. Finally, the soil's parent material is under the B horizon forming C horizon (Hillel, 2004).

Soil is a heterogeneous, polyphasic, particulate, disperse, and porous system. All three ordinary phases in nature are represented in the soil: gaseous, liquid, and solid phases. The gas-phase is the soil atmosphere or air in the soil. The liquid-phase is the water in the soil, which may contain dissolved substances. The solid-phase consists of organic and mineral particles (Hillel, 2004).

Natural earth materials contain varying amounts of internal space. This space is due to the presence of pores generally interconnected, allowing movement of water, air or other liquids and gases through the material, as well as associated chemical species (Thibodeaux, 1996). Soil pores vary in size, quantity, shape, and continuity, and are used to characterize soil structure. Air in soils occupies pores empty of water, and after withdrawal of water, air fills the empty space.

There are many environmental pollutant sources worldwide, varying in their strength, location, and type. The placement of chemicals on and within soil can occur by different processes, for instance application of agricultural chemicals such as pesticides or fertilizers, land disposal for land filling or cultivation, spills, among others (Cohen, 1998). These chemicals move through the soil associated to any of its three phases. Therefore, at first all three phases should be considered when modeling the fate of chemicals in soils.

The objective of the present study is to focus on the solid phase, and mechanisms that enhance the movement of soil particles, as well as substances retained into them. The main three transport/mixing processes that influence the vertical distribution of chemicals sorbed to soil particles are bioturbation, cryoturbation, and movement into cracks formed due to soil drying.

Information in the published literature on these processes constitutes the data collected for this thesis. It is cataloged, tabulated, transformed, theoretically reviewed and evaluated, so as to yield quantitative information needed by chemical fate and transport modelers. Some key results include numerical estimates of sorbed phase soil diffusion coefficients and the effective depths of influence.

Bioturbation, the most significant and most studied of the latter three processes, refers to the disturbance of soil or sediment layers by biological activity. Some species disturb the soil by burrowing and feeding, enhancing the transport of chemicals in this compartment. This process is thoroughly explained in the next chapter.

Cryoturbation is the process of stirring, heaving, and thrusting of soil material resulting from frost action, characteristic of areas at high latitudes with cold arctic or alpine climate. It encompasses frost heave, thaw settlement, and differential mass movements, which are responsible for downslope soil movement in these areas. The extent of cryoturbation features in high altitude areas depends on the amount of available moisture, the rate of freezing, and the types of rocks and soils present in a given area (Benedict 1970)

Freezing and thawing profound influences the stability, hydrology, chemistry, biology, and ecology of soils. Chemicals within the soil profile are redistributed due to the presence of temperature gradients and non-uniform freezing (Lal et al., 2004). Freeze-thaw cycling can also

cause cracks in a soil system, which may open further during subsequent cycles (Richardson, 1976).

Freeze-thaw rates vary with locations. For example in northwest Greenland soils thaw completely in the summer and freeze completely in the winter. Therefore the active part of the year occurs in fall and spring when soils may freeze and thaw on a diurnal basis, probably forming soil cracks. This is different than for example alpine areas, where freezing and thawing may occur on a daily basis for longer periods of time (personal talks, Jennifer L. Horwath)

Due to the presence of water, cracks are also formed in soils, such as clayey soils or vertisols, with high shrink-swell potential (Dasog et al., 1993). When a body of clay sorbs water and dries, shrinking and swelling occur forming numerous cracks. The ability of a soil to crack during shrinking or drying influences many of the transport processes that occur in the soil profile (Horgan et al., 2000). Typical cracks in vertisols are at least 1 cm wide and reach depths of 50 cm or more (Lal et al., 2004). Shrinkage cracks expose considerable hidden subsurface soil. This shrink-swell behavior is also termed dryoturbation.

Consequently, cryoturbation and dryoturbation, similarly to bioturbation, contribute to macropore flow and to the transport of sorbed chemicals downward and upward through the soil. There is an additional soil turnover process that should be considered. It can be described as bioturbation by human activities, and corresponds to the stirring of soil due to agricultural practices, such as plowing, harrowing, disking, among others. Since soil is mechanically mixed by these activities, this process will be called mechanical-turbation. It will be included along with bioturbation.

2. THE SOIL BIOTURBATION PROCESS

2.1 Definition, Contributors, and Effects of Bioturbation

Bioturbation is the turnover or mixing of soil by animals. Some common definitions include, the biologically driven mixing of materials in the soil layer between the underlying geological formations and the overlaying atmosphere (Smallwood et al., 1998), and “the churning and stirring of sediment by organisms” (Bates & Jackson, 1984). Some authors (Eldridge, 2002, 2004; Whitford & Kay, 1999) also refer to soil disturbance by animals as biopedturbation.

Effects of bioturbation are evident in the upper soil horizons O and A. Often equivalent to the A-horizon or topsoil is the biomantle. It constitutes the upper part of soil produced by biota, essentially by bioturbation, and may include deeper levels in some soils. Since organisms bioturbate differently, and innumerable species are involved, the formational pathways of biomantles are exceedingly complex, and vary widely from place to place (Johnson et al., 2003).

Soils accommodate an extensive variety of biota. Although not generally visible to the naked eye, it is one of the most diverse habitats on earth and contains one of the most diverse assemblages of living organisms (Giller et al., 1997). “Nowhere in nature are species so densely packed as in soil communities” (Hågvar, 1998). Soil provides a range of habitats for a multitude of fauna ranging from macro- to micro- levels depending on climate, vegetation, and physical and chemical characteristics of the given soil. Species numbers, composition, and diversity depend on many factors including aeration, temperature, acidity, moisture, nutrient and organic matter content (FAO).

The easiest and most widely used classification system for soil biota divides them into three main groups, based on body size: micro-, meso-, and macrofauna. Microfauna include

nematodes, protozoa, bacteria, etc. These are the smallest organisms (<0.1mm in diameter), extremely abundant, ubiquitous, and diverse. Mesobiota organisms range in size from 0.1 to 2 mm in diameter, including mainly microarthropods (insects, crustaceans, arachnids, etc.). Macrofauna are organisms generally greater than 2 mm in diameter and visible to the naked eye. These are large enough to disrupt the structure of mineral and organic soil horizons through their feeding and burrowing activities (Anderson, 1988). The macrofauna group is the most mobile fauna, moving through macro- and micro-pores in the soil. They include vertebrates (snakes, lizards, mice, squirrels, badgers, armadillo, prairie dogs, crawfish, and others) that primarily dig within the soil for food or shelter, and invertebrates that live in, feed in or upon the soil, the surface litter and their components (ants, termites, millipedes, centipedes, earthworms, snails, spiders, scorpions, crickets and cockroaches).

The macrofauna, maybe termed “ecological engineers”, because they play an important role in moving parts of the soil profile around and form many sorts of burrows and pores (Coleman, 2001). Impacts on soil by animals can be grouped under different processes, such as mounding; mixing (bioturbation); forming and back-filling voids; and regulating soil erosion, movement of water and air in soil, plant litter, animal litter, nutrient cycling, and biota (Hole, 1981).

The mixing of soil is involved in the construction of mounds produced by the superficial deposition of materials from within the soil body. Animals move through the soil to obtain nutrients and water, or to seek protection from predators or environmental variability. In doing so, they penetrate the soil vertically and horizontally, having strong direct influences on the soil (Gabet et al., 2003). The construction of mounds and the mixing of upper soil layers by animals lead to the formation of voids. These voids can be made by animals that excavate, named fossers,

like moles and gophers. Miners, including termites and ants, seize sand grains or bite off fragments of aggregates and bring them to the surface. Tunnelers push or eat their way through soils, like earthworms, snakes, etc. (Hole, 1981).

Bioturbation occurs in different ways, depending on the species involved and the way they move the soil. Some animals feed on biomass produced by plants and organic matter bellow ground; others conduct most of their activity above ground, but live in dens constructed below it. Invertebrates like earthworms, that live underground and move through the soil by pushing particles aside, or like termites and ants, that excavate large quantities of soil, physically alter the soil. In addition, those animals that consume organic matter, like gophers that displace soil while burrowing to eat plant roots, affect the soil structure and biogeochemistry (Gabet et al., 2003).

Burrows or tunnels made by animals increase soil porosity, promote aeration of the soil, and increase infiltration of water. The effects of these burrow systems depend upon the depth and length of the burrows (Whitford & Kay, 1999).

Earthworm cast, a pile of earth egested by a worm, deposition generally decrease the rate of typical soil processes, such as decomposition and biochemical reactions (Edwards et al., 1998), and also promotes the vertical mixing of soil inhibiting soil profile formation. The same effects occur with mounds deposited on the surface by mammals. Ants' burrows and termites' galleries below ground appear to decrease soil bulk density, in an effect similar to that found by Schaefer and Sadlier (1981) with burrowing mammals.

The influence of animals activity on water infiltration rates in soils has been recognized by several authors (Anderson, 1982; Eldridge, 1993; Eldridge et al., 2002; Gabet et al., 2003; Whitford & Kay, 1999; Lal, 1988), concluding that infiltration capacity is enhanced by soil biota. Also, there is an increased depth of water penetration. The construction of channels increases

porosity and particularly those with openings at the surface augment the movement of water through the soil. Additionally, soil nutrients, rates of litter decomposition, mineral concentration, and rates of erosion tend to increase (Whitford & Kay, 1999)

According to Anderson (1988), invertebrates can affect the transport of organic and inorganic materials in soils. He showed that a small biomass of soil macrofauna can have significant effects on fluxes of organic matter and dissolved materials in the superficial horizons of forest soils. Body size and feeding behavior of fauna influence these processes. An example is the movement of soil materials through cast deposition, and litter incorporation by earthworms. Transport of dissolved materials is affected due to the alteration of water movement on and in soils. Distribution of soil particles is also altered through bioturbation; therefore transport of sorbed materials is affected as well.

Environmental impacts of animal burrowing in hazardous waste management systems have also been studied. Bioturbation has been associated with upward movement of radionuclides, and it is the most likely explanation for the widespread radiological contamination on surface soils. At the same time, there is transport of contaminated soil downward when entrained in animal fur or ingested and then excreted below ground (Smallwood et al., 1998).

Figure 2 illustrates an example of how deposited materials can be covered by soil mounds excavated from animal burrows, changing hazardous waste profile in soils.

Similar to the parameters given by Smallwood and others (1998) to determine the impacts of burrowing animals on the risk of environmental exposure from chemicals at waste management sites, the following are helpful to estimate the impacts of bioturbation on the fate of chemicals and their distribution in soils:

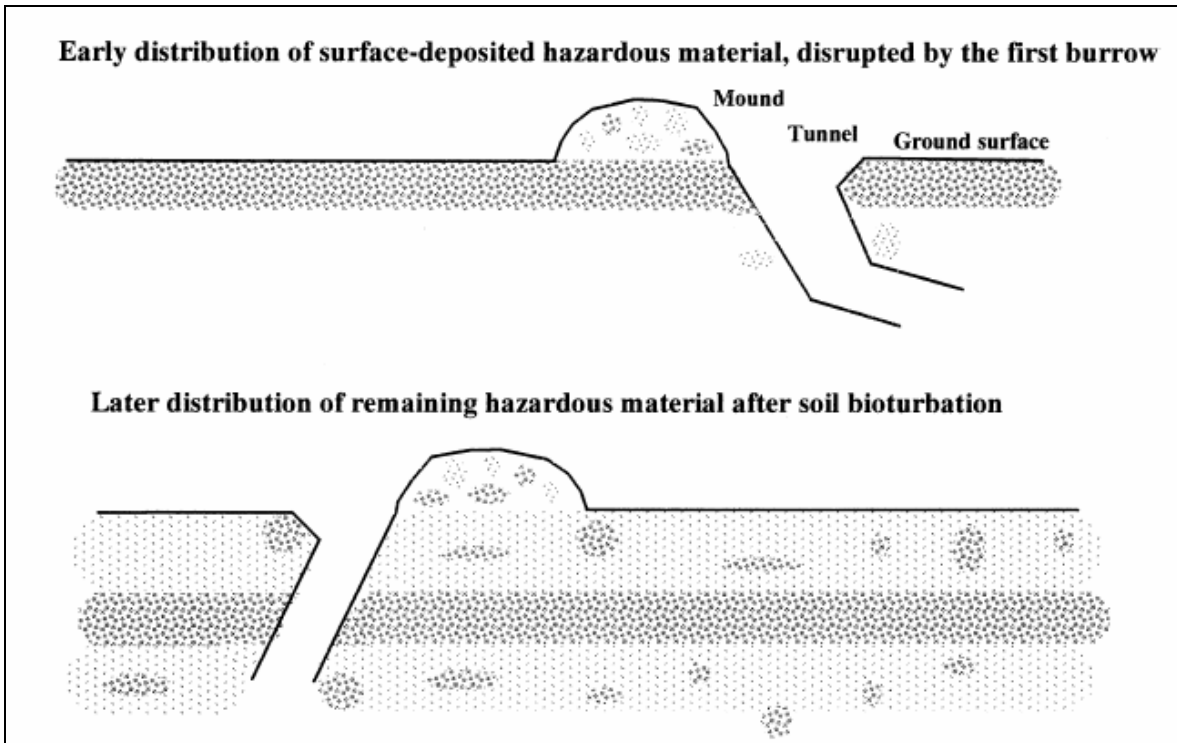


Figure 2. Change in hazardous waste profile in soil due to bioturbation (from Smallwood et al., 1998)

- catalog burrowing animals that occur at the soil site
- estimate abundance or biomass per unit area
- determine maximum and average burrow depths
- estimate rates of soil turnover
- identify characteristics of soil such as porosity, organic carbon content, density, texture, water content, etc.

2.2 Soil Fauna Description

As previously mentioned soil-inhabiting organisms are grouped according to their size in micro-, meso-, and macrofauna, with the latter group being most responsible for bioturbation. Therefore, in order to understand the magnitude and effects of soil bioturbation, it is necessary to describe some representative fauna in this group.

2.2.1 Invertebrates

Most common and known groups of macrofaunal invertebrates that live in the soil are earthworms, ants, and termites. The number of individuals is staggering, ranging from tens to tens of millions/m² (Gabet et al., 2003).

Earthworms are large, abundant, and active; hence their effects on soils have been widely studied. Darwin (1881) was one of the first who recognized the important role of earthworms in affecting soil structure, organic matter processing, and nutrient cycling. He observed and carefully studied earthworms' habits, showing how they cause surface objects to migrate downward by ingesting soil at depth and regularly depositing it upon the surface (Johnson, 1999)

The effects of earthworms on soil processes can vary with different species as well as with different soil types, climatic regimes, etc. Earthworm species have different habitat and feeding preferences which can lead to varied impacts when they invade (Hale, 2005b). Based primarily on their feeding and burrowing strategies, three major ecological groups of earthworms have been defined (Bouché, 1977). Epigeic species are generally small bodied (~3-9cm) and live in or near the surface litter, feeding on litter and organically enriched surface layers of soil (Hendrix, 1995; Hale 2005b; Hendrix et al., 2002). Endogeic species vary considerably in size (~2-11cm in length) and live within the soil profile, in the mineral soil horizon (~0-40cm). They feed primarily on soil and associated organic matter, and form persistent lateral branching burrow systems (Hale, 2005b) with some vertical components and some openings to the surface. Anecic species are large bodied earthworms (>10cm in length) that burrow deeply into the soil horizon (up to 1-2 m deep). They live in more or less permanent vertical burrows that may extend several meters into the soil profile, and feed primarily on fresh surface litter (Hale, 2005b; Hendrix, 1995). These deep burrows enable anecic earthworms to select the conditions that suit

them best from the range of microenvironments available in soil horizons (Lee, 1985). Table 1 summarizes these ecological categories and characteristics.

Table 1. Ecological categories, habitat, feeding, and size of earthworms

<i>Category</i>	<i>Habitat</i>	<i>Food</i>	<i>Length (cm)</i>
Epigeic	Litter	Leaf litter, microbes	< 10
Endogeic	Upper 0-40 cm of soil	Soil with high organic content	~ 2 – 11
Anecic	Burrows	Litter and soil	> 10

(Modified from Hendrix et al., 2002)

Figure 3 illustrate a deep vertical burrow, and some horizontal burrows close to the surface, usually filled with leaves, mineral matter from the surface, or casts.

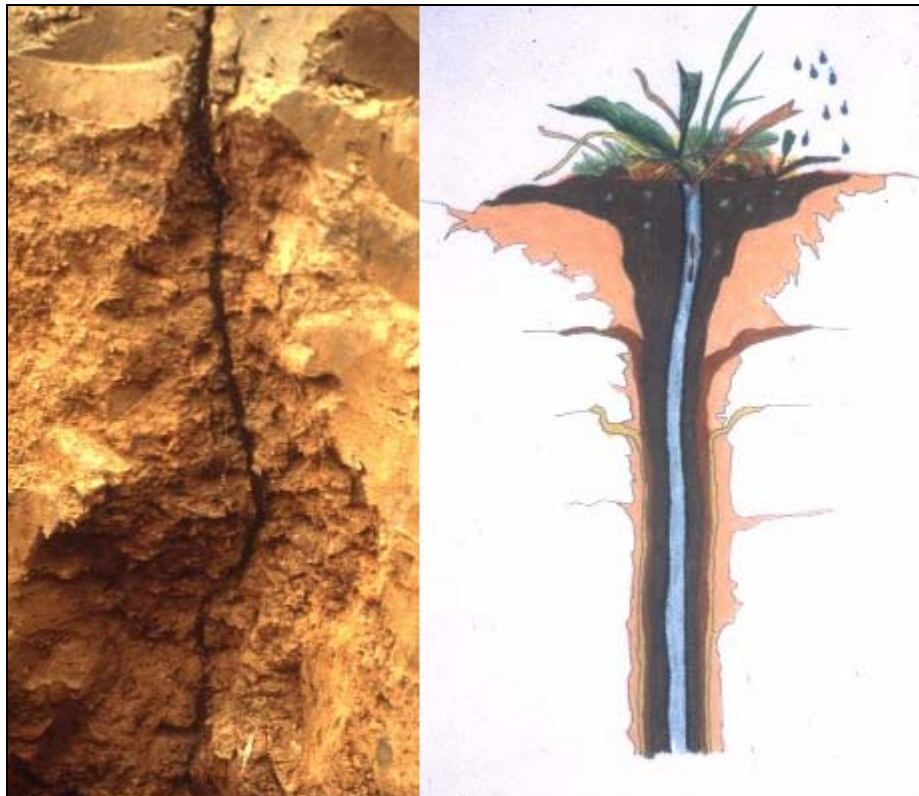


Figure 3. Earthworm permanent burrows (adapted from SWCS, 2000)

Earthworms are the best known and, in many situations, the most important animals that live in soils. Worldwide, over 3500 species have been described (Hendrix et al., 2002) divided into 23 families and several genera. Although not numerically dominant, earthworms' large size makes them one of the main contributors to invertebrate biomass in soils, and therefore to

bioturbation. They are found in most regions of the world, except those with extreme climates (Edwards, 2004), the driest and the coldest land areas. Table 2 shows just a few of the several thousands of species classified with respect to their ecological groups.

Table 2. Selected earthworm species

	<i>Epigeic</i>	<i>Endogeic</i>	<i>Anecic</i>
Species	<i>Dendrobaena octaedra</i>	<i>Aporrectodea spp.</i>	<i>Lumbricus terrestris</i>
	<i>Dendrodrilus rubidus</i>	<i>Octolasion spp.</i>	<i>Allolobophora spp.</i>
	<i>Lumbricus rubellus</i>	Diplocardia spp.	
	<i>Eisenia fetida</i>		
	<i>Eudrilus eugeniae</i>		

One of the most important and most studied families of earthworms is Lumbricidae, which includes the genera *Lumbricus*, *Aporrectodea*, *Bimastos*, *Dendrobaena*, *Eisenia*, *Dendrodrilus*, *Octolasion*, and several others. They are original from Europe, and have been transported by human activities to many parts of the world (FAO). In the United States the only known native representatives of this family fall into two genera, *Bimastos* and *Eisenoides*.

Among several others, Diplocardia species from the Megascolecidae family are also known Nearctic earthworms. Apart from the 100 or more native species, there are at least 45 exotic species that have been introduced in North America, including European Lumbricidae; African, Asian, and South American Megascolecidae; African Eudrilidae, etc. (Hendrix et al., 2002)

Worms excavate their burrows in two ways, by pushing away soil particles and by swallowing them, usually emptying their gut on the surface (Darwin, 1881). In this way there is a continuous turnover of the soil, which will be different depending on the earthworm size, species and its ecological group. According to studies done by Hale, Frelich, and Reich (2005a) in northern forest of Minnesota, each group of earthworms has different potential to remove the forest floor and is likely to have different impacts on the ecosystem as a whole, due mainly to

their different habitat and feeding preferences. The impact of earthworms on soil horizon thickness has also been studied, finding that increasing earthworms' biomass was associated with declining upper O horizons thickness and increasing A horizon thickness (Hale et al., 2005b).

Ants are also important invertebrate soil inhabitants that could be placed in either an anecic or epigeic group (de Bruyn, 1999). Lavelle and Pashanasi (1989) found in a range of land management types the proportion of ant biomass to be minor compared with other soil macrofauna. However, in terms of population density, ants make a more significant contribution.

Similar to earthworms, ants' abundance and their effects in soils depend on factors such as climate conditions, soil type, moisture, land cover and management, among others.

Bioturbation by ants has important implications for soil formation rates and the redistribution of soil particles, nutrients, and organic matter. Analogous to earthworms, ants make complicated burrows deep into the soil, but contrasting with them, ants construct their nests by pulling out soil particles, and carrying loosened soil to another place where they deposit it (de Bruyn, 1999). Their complicated and extensive burrows form a network of macropores underground, mainly concentrated near nest openings, making hydraulic conductivity highest in these areas (Gabet et al., 2003).

In arid and semi-arid areas where vegetation cover is often low, ants and termites play an important role in bioturbation, movement of soil particles, and infiltration. Eldridge (1993) observed nest entrances constructed by funnel ants in semi-arid woodland in Australia surrounded by a raised crater of soil called torus. These holes are usually filled with litter, particularly leaves, from the entrance contributing to soil turn over. Studies done by Eldridge and Pickard (1994) revealed that funnel ants transport large amounts of soil to the surface (see rates in next section); suggesting that over time this soil will lead to the development of a new layer.

Ants interact with soil processes increasing soil porosity and infiltration, reducing bulk density, regulating soil erosion, and concentrating organic matter around their nests (Hole, 1961).

Termites have many of the same effects on soils as ants. They excavate large galleries below ground and construct surface mounds, altering soil profile. Similar to other invertebrates, termites play a significant role in soil formation through soil turnover and physical disturbance, increasing macroporosity and therefore water infiltration (Lal, 1988).

Termites are often considered the tropical analogs of earthworms since they reach large abundances in the tropics and process large amounts of litter. The principal difference is that earthworms egest much of what they ingest in altered form whereas termites transfer large amounts of soil and organic material into their nests and mounds (Coleman, 2001).

These feeding and burrowing activities of earthworms, ants, termites, and other soil invertebrates influence the physical structure of soils, water flux pathways and hence the transport of materials. These effects can be considered as modifications to the basic patterns of water and chemicals movement in soils (Anderson, 1988).

2.2.2 Vertebrates

Many species of mammals burrow into and through the soil. Most excavate dens that serve as protection when they are not active. These dens can be extensive, and usually stable through time (Gabet et al., 2003). Rodents are the most important vertebrates with substantial impact on soils, including pocket gophers, kangaroo rats, ground squirrels, and prairie dogs. (Hendricks, 1985)

Gophers occur over much of the North America continent west of the Mississippi River and in the Southeast. Their impacts on soil can be profound. Rodents, like invertebrates, excavate long burrows and place the loose soil on the surface as mounds or deposit it into abandoned

burrows. The net effect of gophers is to mix soil vertically and generate irregular soil conditions horizontally.

To illustrate the magnitude that bioturbation by vertebrates can reach, it is worthy to mention one hypothesis proposed to explain the formation of Mima mounds. Generally found in western United States, the creation of these circular mounds of dirt, which can reach up to 2 m high with a diameter of 25-50 m, has been attributed to fossorial rodents (Gabet et al., 2003). Cox (1984) has presented convincing evidence that pocket gophers maintain, and may create, Mima mounds. Johnson, Johnson, and West (1999) examined Mima mounds in Iowa and Minnesota, and stated that pocket gopher bioturbation was active on all mounds.

In the western United States, American badgers are a major predator of ground squirrels and other burrowing fauna. While preying, badgers enlarge the small squirrel holes producing a large mound at the entrance. A study conducted by Eldridge (2004) demonstrated that badgers produce extensive soil disturbance that initiates notable changes in landscape structure. The holes excavated by them become sinks for soil, seeds, and litter.

The effects of bioturbation by several mammals in different environments have been recognized by scientists from diverse fields. According to Whitford & Kay (1999) excavations by animals such as wombats, kangaroo rats, and pocket gophers are important in bioturbation evolution and redistribution of materials in soil profiles. In deserts, small mammals' movement may be the most important mechanism for pumping soluble nutrients from deep soil layers on the surface and may be the only mechanism for bringing insoluble materials to the surface. Prairie dogs are an additional example of these mammals; they burrow extensively and develop substantial mounds around their burrow entrances. These abandoned mounds contribute to soil mixing and their burrows serve as litter and seed traps. (Whitford & Kay, 1999).

Resting forms and beds of intermediate size mammals may embody important bioturbation features (Whitford & Kay, 1999). In Australia, kangaroos excavate pits termed hip holes as resting sites. Eldridge and Rath (2002) found hip holes dimensions of 82 cm long by 58 cm wide by 9 cm deep. According to the volume of soil removed by these kangaroos, they should be recognized as significant controllers of ecosystem processes. A major effect of kangaroos is to destroy soil structure and therefore aggregate stability through digging and scratching.

2.3 Agriculture and Mechanical-turbation

The effects of soil organisms previously mentioned are also relevant for agricultural purposes. As soil species feed on organic matter, nutrients become available for plant use. Their activity improves soil structure by increasing soil porosity and aggregation, promotes aeration, reduces compaction, and increases water movement. Therefore, creating a favorable environment for living organisms in soils emphasizes plant growth and reduces maintenance (Whiting et al., 2005).

Apart from actions and effects of living organisms in soils, human activities should be considered as having important repercussion in this living ecosystem. Even though humans do not live in soils, they make use of land for a variety of purposes, agriculture being one of the most important uses.

Replacement of a natural ecosystem by an agro-ecosystem entails changes in topsoil properties. The agricultural preparation of land to receive seeds, known as tillage, involves mechanical actions exerted on soil that modify soil conditions using various combinations of equipment, such as conventional moldboard plough, disk plough, harrow, hoe, tillers, among others. After intensive tillage, the natural ecological cycle of soils is altered. In prairies, the

annual cycle of grasses creates a deep layer of litter, which protects the soil from erosion and temperature extremes. Soil organisms thrive in the layers of dead grasses developed each season. As prairie plants decay, nutrients return to the soil. Water, instead of running off, seeps back into the soil, replenishing groundwater and nearby streams (Fawcett et al., 2003). This cycle is radically altered by tillage practices.

Recent progress in soil tillage systems has shifted toward less intensive soil cultivation technologies oriented to ecosystem protection. Soil cultivation practices are usually classified according to their impact and distribution of previous crop residues in soil. Conventional tillage, also called intensive tillage, comprises all tillage types that leave less than 15% of crop residues on the soil surface after planting the next crop. Conservation tillage is any tillage system with 30% or more residues remaining after planting (El Titi, 2003).

Following is a brief description of some of these tillage practices. Although this study is not aimed to focus on agricultural activities, it is important to include them as they mechanically mix the soil. This type of mixing will be referred to as mechanical-turbation.

A deep-tillage operation is often required to loosen and break the soil. The implements used include moldboard, chisel, and disk plows. The moldboard plow is categorized as an inversion tillage type. It inverts the topsoil and moves it into deeper soil layers. This inversion incorporates residue from previous crops into the soil, reduces the prevalence of weeds, and makes the soil more porous. Also, the biota of upper soil layers becomes part of the subsoil environment. The moldboard plow has been one of the most widely used implements for many centuries. It cuts and turns topsoil at various depths, typically between 10 and 35 cm, and once a year (El Titi, 2003). However, depending on the location, soil, and crop, it can be done twice, or three times in rare cases.

The use of a chisel plow is considered a noninversion and limited-tillage farming practice. This tool is used to get deep tillage with limited soil disruption, and its main function is to loosen and aerate soils. It is typically set to run up to a depth of about 30 cm. There is a condition of the soil called hardpan in which soil grains become cemented together by bonding agents such as iron oxide and calcium carbonate forming a hard and impervious mass. The chisel plow can also be used to reduce the effects of compaction and to help break up hardpan.

Harrowing is used to cultivate the surface of soils, differing from the plow that is used for deeper cultivation. Plowed land is usually harrowed to pulverize clods of earth and level the soil. It can be an effective method for spreading straw (El Titi, 2003), and also to pull up weeds, aerate the soil, and cover seeds. There are several types of harrows depending on the intended purpose. A common one is the disk harrow applied in the top 5 cm of the soil.

Over the past four decades many farmers are applying conservation tillage. Instead of plowing and disking before planting, they leave the residue of the previous crop on the soil surface. The most effective soil-conserving system is no-till (Fawcett et al. 2003). Several years without tillage are needed to maximize benefits such as reduced soil erosion, improved soil structure, and increased infiltration. However, there are places like northern Ohio where the weather and soils can be cool and wet during the early part of the growing season, where no-till may cause serious problems for certain crops (Comis, 2004).

Conservation tillage includes no-tillage, ridge-tillage, mulch tillage, among others. With ridge-tillage, the soil is left undisturbed except for nutrient injection. With mulch tillage, soil is disturbed prior to planting with tools such as chisels, field cultivators, or disks (El Titi, 2003).

A change in agricultural land use affects nearly every component of the soil ecosystem, including organisms that utilize the soil as habitat. Burrows created by earthworms or other

species can be altered or destroyed after mixing or complete turn over of the soil when plowing or harrowing. Abundance and prevailing species could also be affected by agricultural activities. House and Parmalee (1985) compared a field with 17 years of no-till cropping with a conventionally tilled field and found from 3.5 to 6.3 times more earthworms in the no-till field.

During tillage earthworms are often brought to the soil surface, increasing their exposure to predation by birds, desiccation, and mechanical damage. However, the degree of physical damage appears not to be significant for most earthworm species, compared to the effects of tillage on the incorporation of surface crop residues, which may otherwise provide food resources and protective cover for epigeic species (El Titi, 2003). Tillage intervention not only affects earthworm communities. Similarly, abundance and diversity of other invertebrates and vertebrates are affected.

Responses of soil fauna to physical changes in soil environments are guided by the specific habitat requirements of the species involved (El Titi, 2003).

2.4 Abundance, Distribution, and Cast Production

In order to assess the impacts of bioturbation in the transport of chemicals in surface soils, it is important to know how much soil is usually moved by the species present at the site of interest. Therefore, density or biomass of soil animals, where are most of the species concentrated, and what depths can they reach, are some of the questions that need to be answered.

Since earthworms are the key species in bioturbation, this section will focus on abundance and cast production only by earthworms. Additionally, data for other soil inhabitants is limited and hard to find.

Tables 3 and 4 show abundance and biomass of different earthworm species in diverse habitats of United States and around the world, respectively. Earthworms' biomass from Table 3 was used to build the probability plot on Figure 4. Two trendlines were added to approximately fit the data points corresponding to biomass in forests soils, and grassland soils. According to Figure 4, there is a 50% probability of having approximately 1.95 g/m² or less of earthworms biomass in forests. Similarly, based on a 50% probability of occurrence, earthworms' biomass in soils can be 3.16 g/m² or less.

Additionally, data published by several authors on earthworm cast production is presented in Table 5.

2.5 Soil Turnover Rates, Depths, and Biodiffusion Coefficients

Cast production by earthworms, as those presented in Table 5, can be used to estimate the rate at which earthworms turnover the soil. To make this calculation it is necessary to know the soil density and its porosity, using the following correlations.

$$v = \frac{n_3}{\rho_b} \quad (1)$$

$$\rho_b = \rho_3 \cdot (1 - \varepsilon) \quad (2)$$

where: $v \equiv$ soil turnover rate (cm/year)

$n_3 \equiv$ cast production (g/cm².year)

$\rho_b \equiv$ soil bulk density (g/cm³)

$\rho_3 \equiv$ density of the soil (g/cm³)

$\varepsilon \equiv$ porosity

Table 3. Abundance and biomass of earthworms from selected habitats in USA

<i>Habitat Description</i>	<i>Location</i>	<i>Earthworm Species</i>	<i>Abundance (number/m²)</i>	<i>Biomass (g/m²)</i>	<i>Reference</i>
Temperate hardwood forest (maple-dominated) with thick forest floor	Arnost Forest, central New York	<i>Aporrectodea tuberculata</i>	12.0	5.9	Bohlen et al., 2004
		<i>Dendrobaena rubida</i>	1.0	0.05	
		<i>Eisenia rosea</i>	7.0	1.3	
		<i>Lumbricus castaneus</i>	-	-	
		<i>Lumbricus rubellus</i>	106.0	35.7	
		<i>Lumbricus terrestris</i>	29.1	26.6	
		<i>Octolasion tyrtaeum</i>	95.0	17.1	
		<i>L. terrestris</i> or <i>L. rubellus</i> ^a	22.0	1.3	
		All species	202.0	46.7	
Temperate hardwood forest (maple-dominated) with agricultural history and thin forest floor	Tompkins Farm, eastern New York	<i>Aporrectodea tuberculata</i>	63.0	27.2	Bohlen et al., 2004
		<i>Dendrobaena rubida</i>	-	-	
		<i>Eisenia rosea</i>	-	-	
		<i>Lumbricus castaneus</i>	16.0	1.6	
		<i>Lumbricus rubellus</i>	-	-	
		<i>Lumbricus terrestris</i>	78.0	95.3	
		<i>Octolasion tyrtaeum</i>	120.0	16.9	
		<i>L. terrestris</i> or <i>L. rubellus</i> ^l	-	-	
		All species	196.0	111.3	
Upland soil with slight/moderate/severe erosion	Piedmont of the southern Appalachian mountains in North- central Georgia	-	60/301/255	0.4/3.3/4.4 ^b	Hendrix, 1992
Plowed upland soil		-	154	2.34 ^b	
No-till upland soil		-	307	3.34 ^b	
Bottomland plowed agroecosystems		<i>Lumbricus rubellus</i> and <i>Aporrectodea caliginosa</i> ^c	200 - 1000	1 - 30 ^b	
Bottomland no-tillage agroecosystems		<i>Lumbricus rubellus</i> and <i>Aporrectodea caliginosa</i> ^c	0 - 600	0 - 15 ^b	
Fertilized grass meadows	Pied. Southern	<i>Diplocardia</i> spp ^c	378	-	Hendrix, 1992
Unfertilized grass meadow	Appalachian Mount.	-	190	-	
Hardwood forest	in North-central GA	Lumbricids spp ^c	573	-	

(table continued)

Habitat Description	Location	Earthworm Species	Abundance (nbr/m ²)	Biomass* (g/m ²)	Reference
Beech-maple dominated hardwood forest, in sandy-dry soils.	Western Great Lakes region. Pictured Rocks National Lakeshore, Michigan	<i>Aporrectodea spp</i> ^d	-	1.9 ^b	Hale, 2005b
		<i>Dendrobaena octaedra</i>	-	0.2 ^b	
		<i>Dendrodrilus rubidus</i>	-	0	
		<i>Eiseniella tetraedra</i>	-	0	
		<i>Lumbricus rubellus</i> (adults)	-	0.1 ^b	
		<i>Lumbricus terrestris</i> (adults)	-	2.6 ^b	
		<i>Lumbricus juveniles</i>	-	1.3 ^b	
		<i>Octolasion tyrtaeum</i>	-	0.6 ^b	
Aspen-fir dominated boreal forest, often in shallow or rocky soils	Western Great Lakes region. Voyageurs National Park, Minnesota	<i>Aporrectodea spp</i> ^d	-	0.4 ^b	Hale, 2005b
		<i>Dendrobaena octaedra</i>	-	0.1 ^b	
		<i>Dendrodrilus rubidus</i>	-	0.1 ^b	
		<i>Eiseniella tetraedra</i>	-	0.1 ^b	
		<i>Lumbricus rubellus</i> (adults)	-	0.5 ^b	
		<i>Lumbricus terrestris</i> (adults)	-	0.8 ^b	
		<i>Lumbricus juveniles</i>	-	0.7 ^b	
		<i>Octolasion tyrtaeum</i>	-	0	
Sugar maple dominated forest	Chippewa National Forest of Northern Minnesota	<i>Aporrectodea</i>	-	0.85 ^b	Hale, 2005a
		<i>Dendrobaena</i>	-	0.07 ^b	
		<i>Lumbricus rubellus</i> (adults)	-	0.78 ^b	
		<i>Lumbricus juveniles</i>	-	2.41 ^b	
		<i>Lumbricus terrestris</i> (adults)	-	0	
		<i>Octolasion</i>	-	0.18 ^b	
		All species	-	7.91 ^b	
Natural grassland	Tennessee	Lumbricidae	13 - 41	3.2 – 7.5	Reynolds, 1970
Deciduous forests	Tennessee	Lumbricidae	2 - 96	1.3 - 14	Reynolds, 1970
Deciduous forests	Indiana	Lumbricidae	14 - 124	26.3 – 280.3	Reynolds, 1972

^a these represent juvenile worms that could not be separated by species ^c predominant species
^b ash free dry mass (g/m²) ^d includes *A. caliginosa*, *A. tuberculata*, and *A. juveniles*

Table 4. Abundance and biomass of earthworms from selected habitats around the world

<i>Habitat Description</i>	<i>Location</i>	<i>Earthworm Species</i>	<i>Abundance (nbr/m²)</i>	<i>Biomass (g/m²)</i>	<i>Reference</i>
Tropical Rain Forest	4725mm/yr rainfall Los Tuxtlas, Mexico	Endogeics ^a	34	9.0	Fragoso et al, 1992
	4015mm/yr rainfall Volcan Barva, Costa Rica	-	340	53.7	Atkin and Proctor, 1988
	3521mm/yr rainfall Rio Negro, Venezuela	Epigeics ^a	55	14.2	Fragoso et al, 1992
	1276mm/yr rainfall Lamto, Ivory Coast	Endogeics ^a	35	1.6	Lavelle, 1978
Tropical Rain Forests	12 different sites	All species (average)	68	12.9	Fragoso et al, 1992
Sown pastures	South Australia	Lumbricidae	460 - 625	62 - 78	Barley, 1959
Pastures with heavy rates of fertilizers	Ireland	Lumbricidae	400 - 500	100 - 200	Cotton and Curry, 1980
Cropland	South Australia	Lumbricidae	20 - 25	2 - 2.5	Barley, 1959
Natural grassland	Wales	Lumbricidae	22	8	Reynoldson, 1966
Tropical savannas	Ivory Coast	Megascolecidae and Eudrilidae	230	49	Lavelle, 1974
Coniferous forests	Japan	Lumbricidae	27 - 72	-	Brauns, 1955
Deciduous forests	Canada	Lumbricidae	240 - 780	38 - 109	Maldague, 1970
Tropical forests	Nigeria	Eudrilidae	61.7	2.5	Cook et al, 1980
Old pasture	France	Lumbricidae	288	125	Bouché, 1977

^a predominant species

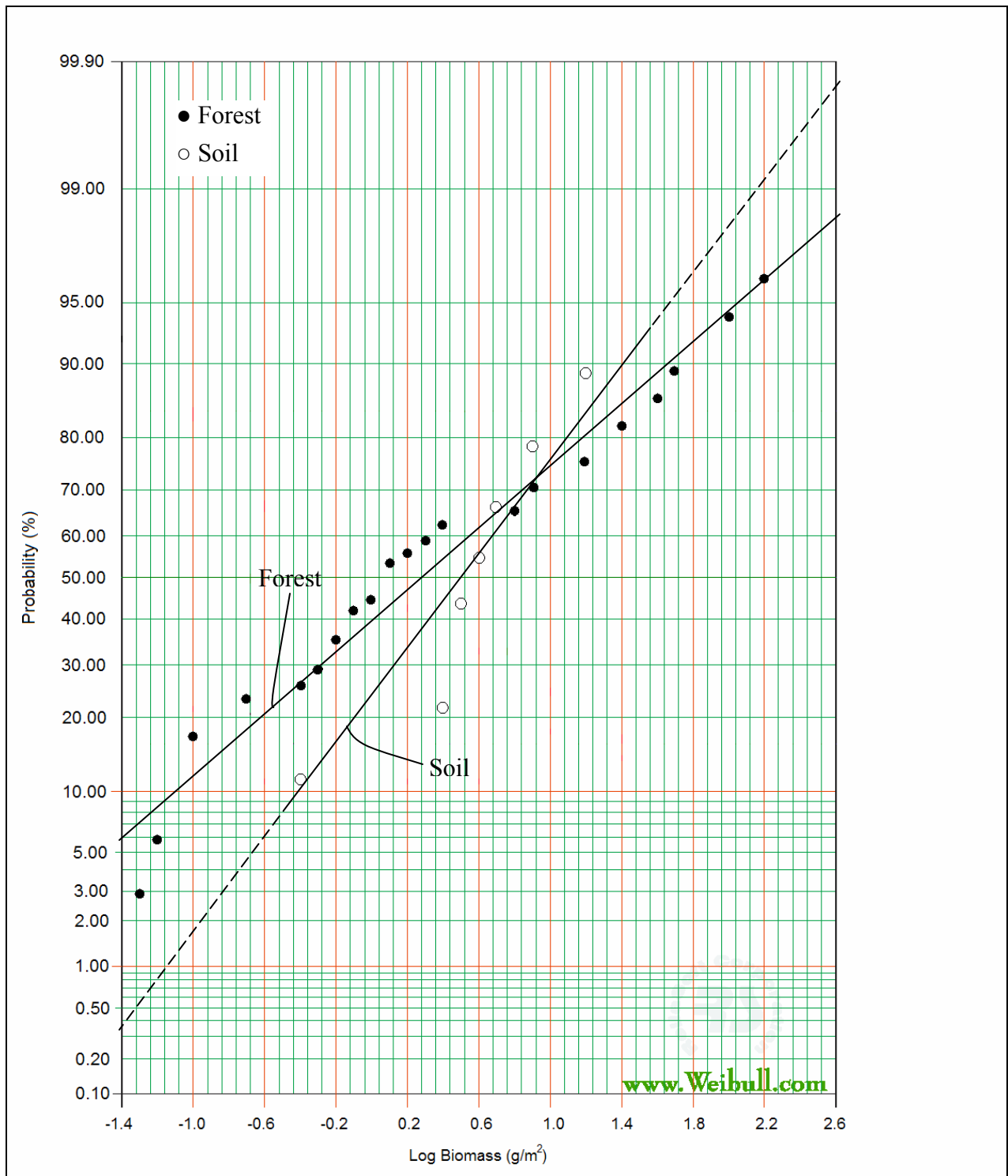


Figure 4. Cumulative probability of earthworms biomass in forest and soil habitats in USA

Table 5. Annual cast production by earthworms in various regions of the world

<i>Habitat Description</i>	<i>Location</i>	<i>Earthworm Species</i>	<i>Biomass (g/m²)</i>	<i>Surface casts (g/m²)</i>	<i>Total casts (g/m²)</i>	<i>References</i>
Tallgrass prairie, with shallow rocky soils	Konza Prairie Research	Diplocardia spp	1.44 ± 1.35 ^a	170	7571	James, 1991
		Lumbricidae	1.39 ± 2.23 ^a	21	1380	
Tallgrass prairie, with deep and downslope soils	Natural Area in Kansas, USA	Diplocardia spp	0.95 ± 0.54 ^a	201	4986	James, 1991
		Lumbricidae	4.43 ± 3.6 ^a	80	4406	
Pasture	France	Lumbricidae	165.4	7000	-	Bouche, 1982
Beech wood Oak wood	Germany	Lumbricidae	-	6800	-	Kollmannsperger, 1934
		Lumbricidae	-	5800	-	
Pasture	Australia	<i>A. caliginosa</i>	62 - 78	250	-	Barley, 1959
Shrub savanna	Ivory Coast	Megascolecidae	-	7100	-	Lavelle, 1978
		Eudrilidae	54.5	20700	-	
Grassland	South Africa	Microchaetus spp	96	5000	-	Ljungstrom et al, 1969
Pasture	Japan	<i>Pheretima hupeiensis</i> <i>Eisenia japonica</i>	7.6	3800	-	Watanabe, 1975
Pot studies	-	<i>A. caliginosa</i>	-	3000 - 4000	-	Barley, 1959
Temperate grasslands	-	-	-	-	50000 ^b	Lavelle, 1988
Pastures	New Zealand	-	-	2500 - 3300	-	Syers et al, 1979
Poor pasture	England	-	-	4470	-	Darwin, 1883
Temperate pastures and grasslands	-	-	-	4000-5000	-	Anderson, 1988
Tropical savanna	West Africa	-	-	120000 ^b	-	Anderson, 1988

^a ash free dry mass (g/m²)^b as high as

The depth of burrows, tunnels, or holes, is highly variable and species specific, depending on several factors which will be mentioned later.

According to Lee (1985) earthworm burrows are frequently close to the surface but most species, mainly the lumbricids that inhabit much of the area of grasslands and croplands, live in semi-permanent burrow systems that penetrate to about 20-50 cm into the soil.

In a comparative study of tropical rain forest earthworm communities, Fragoso and Lavelle (1992) found that earthworms generally occur at depths of 0-40 cm, mainly concentrated in the upper 10 cm. In a deciduous forest soil in Wisconsin, *Lumbricus terrestris* commonly pull leaves into their burrows to a depth of 10 cm, within burrows that are up to 1 m deep (Hole, 1981). In another study done with contaminated soil columns, Singer et al. (2001) found that earthworms' activity was greatest within the 2-6 cm depth. In a permanent pasture at Waite Agricultural Research Institute in Australia, the maximum depth of burrow found was 25 cm by *Aporrectodea rosea* and *A. caliginosa* (McKenzie et al., 1993). In the same location, a mixed earthworm population made vertical burrows approaching 1 m, attributed to *L. terrestris*, and twisting burrows down to a maximum of 27 cm, attributed to *A. caliginosa* and *A. rosea*.

Depth of earthworm burrows have been measured by several authors, as mentioned above. In agroecosystems considerable differences are evident with differing agricultural practices. For example, Ehlers (1975) compared corn fields in Germany that had been subject to normal tillage with fields where no tillage was applied for four years, showing a great increase in earthworms' density where zero-tillage methods were practiced, and finding burrows from 2 to 60 cm deep. In croplands in USA Hopp (1973) found depth burrows of 7.5 cm.

Ants and termites tunnel depths are comparable to those of earthworms. Hole (1981) reports that the ant *Formica neogagates* mixes the upper 35 cm of the soil. Eldridge (1993) also found ant tunnels at depths ranging from 10 to 35 cm.

Generally, vertebrate burrows occur from 10 – 30 cm bellow the surface with dens extending down to approximately 1 m (Gabet et al., 2003). According to Eldridge and Rath (2002) kangaroo hip holes can reach depths of 10 cm or more.

The process of bioturbation can be mathematically described with a bioturbation diffusion or biodiffusion coefficient that can be estimated using soil turnover rates and depths reached by soil organisms. Starting with the concept of probability of random particle displacement over time applied in general molecular diffusion processes, the soil particle displacement based biodiffusion coefficient can be obtained as follows:

$$D_3 = \frac{\bar{x}^2}{2t} \quad (3)$$

where: $D_3 \equiv$ biodiffusion coefficient (cm^2/year)

$\bar{x}^2/2 \equiv$ particle mean-square displacement (m^2)

$t \equiv$ diffusing time (year)

Rewriting equation 3 gives

$$D_3 = \frac{\bar{x}}{t} \cdot \frac{\bar{x}}{2} = v \cdot h_3 \quad (4)$$

where $\bar{x}/2$ correspond to the vertical displacement of the particles, equivalent to the average depth of burrows or tunnels (h_3), and \bar{x}/t denotes the soil turnover rate by animals. Equation 4 represents the concept of a characteristic velocity and a characteristic path length

used by Mclachlan, Czub, and Wania (2002) to obtain approximated bioturbation diffusion coefficients.

Analogously, a diffusion coefficient can be calculated to model disturbance of soil due to agricultural practices. In this case the soil turnover rate can be calculated as follows

$$\frac{\bar{x}}{t} = \frac{V \cdot s}{A} = \frac{A \cdot h_3 \cdot s}{A} = h_3 \cdot s \quad (5)$$

where: $V \equiv$ soil volume turned over (cm^3)

$A \equiv$ soil area (cm^2)

$s \equiv$ frequency of plowing (times per year)

$h_3 \equiv$ depth of plowing layer (cm)

Therefore, replacing Equation 5 into 4 gives a diffusion coefficient for mechanical-turbation

$$D_3 = h_3 \cdot s \cdot \frac{h_3}{2} = \frac{h_3^2 \cdot s}{2} \quad (6)$$

Equations 4 and 6 can be used to estimate soil biota and plowing biodiffusion coefficients.

2.6 Bioturbation Related Factors

From the data in Tables 3, 4, and 5, it can be noticed that population abundance, biomass, and annual cast production are highly variable depending on the animal, habitat, and location, aspects on which soil turnover rates and depths also depend. At this point it is important to mention that the rate of mixing of soil declines with depth (Humphreys et al.), probably because the number and density of species also falls off with depth.

The size of earthworm populations and distribution depends on a wide range of factors, including:

- Soil properties: texture, depth, moisture-holding capacity, pH, density, organic matter content.
- Ecosystem: land cover, location, soil management methods.
- Species: size, feeding preferences, habitat tolerances, interaction with other species, burrowing habits.
- Climatic regimes: temperature, rainfall, seasonal variability.

The combination of all these factors characterizes bioturbation, not only for earthworms but for soil biota in general. Therefore, it is difficult to determine or predict soil turnover rates for any species or habitat of interest. In fact, data available in different habitats and soil types is spotty and very incomplete, especially for natural ecosystems.

For example, in the case of earthworms, warmer climates with longer growing seasons may have more soil turnover, but there is an upper limit. Once it gets too dry or hot worms will become inactive. In Minnesota, there is about a 5-month season in which earthworms are active (May - September), but they are often inactive in parts of July and August due to high temperatures and dry soil conditions, which can also vary from year to year (Hale, personal talks). In temperate regions, earthworms are more active in spring and fall. During winter, they retreat to the deeper soil layers to escape adverse temperature conditions, though they can become quite active again during cool periods when the ground is not frozen (Hendrix, 1995).

The previous factors related to bioturbation could be grouped into two main aspects, which are important in assessing and estimating the impact of bioturbation on the transport of chemicals in soils and can be used in CFaT models. These are:

- Geographic climatic region
- Species type

There are several systems for classifying the world's climates based on the patterns of temperature and precipitation, a widely known one is Köppen classification (Pulsipher et al., 2002). A geographic climatic region with the same temperature and rainfall conditions would encompass similar land cover or vegetation, soil properties, and seasonal variability.

Some scientists use a land-based ecoregion classification. In this system, each terrestrial ecoregion is distinguished by its shared ecological features, climate, and plant and animal communities (NGS). Once a region type is assigned to an area and the species present are known, it is possible to characterize bioturbation.

2.7 Results from Literature Review

Table 6 summarizes soil turnover rates calculated using Equation 1 and the surface casts' production by earthworms presented in Table 5. A soil bulk density of 1.3 g/cm^3 was assumed. Also, it includes rates reported by Darwin (1881). These are conservative soil turnover estimates because they are based on surface cast production, without considering casts deposited within the soil profile.

Soil turnover rates by other invertebrates and some vertebrates are presented in Table 7.

2.8 Presentation of Results and Discussion

2.8.1 The Bioturbed Depth

The information presented in the previous section on depths for the various species studied has been plotted in Figure 5 for an easier visualization and for future use. It shows depths “definite” to be reached and those reported depths “likely” to occur. The definite designation refers to the surface and adjoining layers contacted by all organisms while the likely designation denotes depths which few organisms have been observed.

Table 6. Soil turnover rates by earthworms in various regions of the world

<i>Habitat</i>	<i>Location</i>	<i>Earthworm Species</i>	<i>Soil Turnover Rate (cm/year)</i>	<i>Reference</i>
Tallgrass prairie	Kansas, USA	Diplocardia	0.015	James, 1991
		Lumbricidae	0.002 - 0.006	
Pasture	France	Lumbricidae	0.5	Bouché, 1982
Forest	Germany	Lumbricidae	0.4 – 0.5	Kollmannsp., 1934
Pasture	Australia	<i>A. caliginosa</i>	0.02	Barley, 1959
Shrub savanna	Ivory Coast	Megascolecidae	0.6	Lavelle, 1976
		Eudrilidae	1.6	
Grassland	South Africa	<i>Microchaetus</i>	0.4	Ljungstrom, 1969
Pot studies	-	<i>A. caliginosa</i>	0.2 – 0.3	Barley, 1959
Pastures	New Zealand	-	0.2 – 0.3	Syers et al., 1979
Poor pasture	England	-	0.3	Darwin, 1883
Temperate pastures and grassland	-	-	0.3 – 0.4	Anderson, 1988
Tropical savanna	West Africa	-	8.5 ^a	Anderson, 1988
Good pasture, no ploughed	England	-	0.56	Darwin, 1883
Stony field	England	-	0.21	Darwin, 1883
Swampy field	England	-	0.48 – 0.53	Darwin, 1883
Dry sandy grassland	England	-	0.56	Darwin, 1883
Pasture, partially planted	England	-	0.51 – 0.56	Darwin, 1883

^a as high as

Table 7. Soil turnover rates by different soil biota species

<i>Species</i>	<i>Location</i>	<i>Habitat</i>	<i>Soil Turnover Rate (cm/year)*</i>	<i>Reference</i>
Ants				
funnel ants	Brazil	<i>not specified</i>	0.31	Weber, 1966
	New South Wales, Australia	Semiarid Aeolian soil	0.03	Eldridge et al., 1994
mixed species	Kellerberrin, Australia	<i>not specified</i>	0.003	Briese, 1982
		Rural environments	0.003	de Bruyn et al., 1994b
<i>lasius niger</i>	Michigan, USA	Naturally vegetated environments	0.002	
<i>formica exsect.</i>	Wisconsin, USA	Rural environments	0.007	Talbot, 1953
<i>formica fusca</i>	USA	Rural environments	0.09	Salem et al., 1968
4 ant species	USA	Naturally vegetated environments	0.54	Wiken et al., 1976
	Chihuahuan Desert, USA	Naturally vegetated environments	0.0007	Whitford, 1996
Termites				
	<i>not specified</i>	Semiarid environments	0.013 - 0.2	de Bruyn et al., 1994
	Senegal	Tropical ecosystems	0.014	Lepage, 1974
	Uganda	Tropical ecosystems	0.01	Pomeroy, 1976
<i>macrotermes</i> spp.	<i>not specified</i>	Oxisol, Ultisol soil	0.09	Lee et al., 1971
	Kenya	not specified	0.008	Anderson, 1988
<i>macrotermes</i> spp.	Africa	Savannas	0.08	Anderson, 1988
Badgers				
american badger	west-central Idaho, USA	Mixed sagebrush vegetation, fine, silty soils	0.04	Eldridge, 2004
	New Mexico, USA	Chihuahuan Desert	0.02	Whitford et al., 1999
Gophers				
pocket gopher	USA	<i>not specified</i>	0.034 – 0.6	Smallwood et al., 1999
pocket gopher	New Mexico, USA	Chihuahuan Desert, grassland community	0.7	Whitford et al., 1999
pocket gopher	USA	<i>not specified</i>	max. 0.8	Cox, 1990
pocket gopher	<i>not specified</i>	<i>not specified</i>	0.15	Hole, 1981
Squirrels				
prairie dog	<i>not specified</i>	Ustoll soil	0.54	Thorp, 1949
arctic ground squirrel	<i>not specified</i>	<i>not specified</i>	0.15	Hole, 1981
Kangaroos				
grey kangaroo	western New South Wales, Australia	Semiarid wooded rangeland	0.02	Eldridge et al., 2002

* calculated assuming soil bulk density of 1.3 g/ml

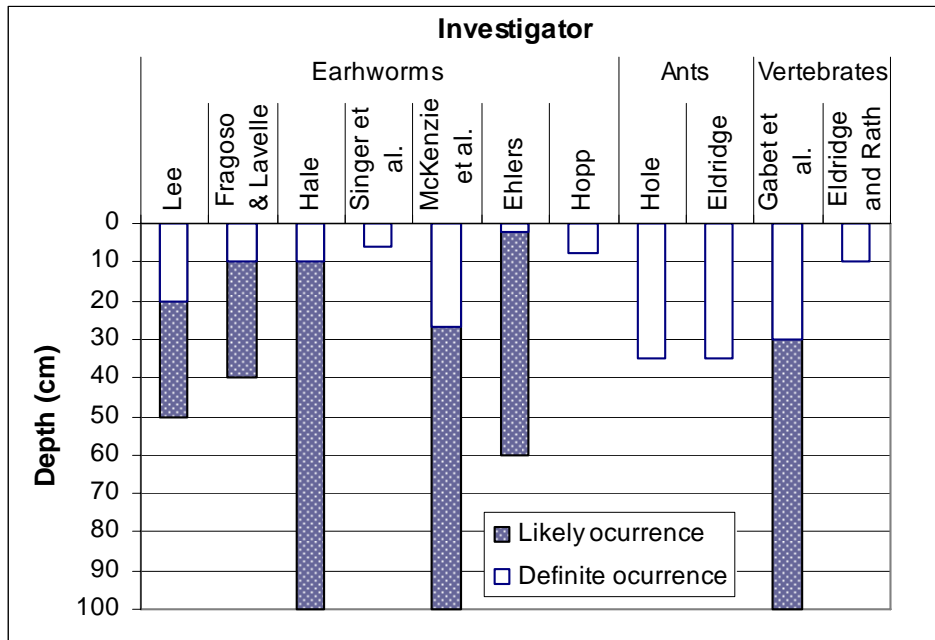


Figure 5. Definite and likely depth occurrences

Based on the previous chart, Figure 6 has been created to show the probability of a species to reach certain depth. It was built counting only the number of definite occurring depths in Figure 5. A straight line was added to Figure 6 to see if a normal distribution fits these data. Since data fall approximately along the line, normality was assumed and used as a guide when modeling bioturbation. Based on 50% probability, it appears that the average depth of soil bioturbation is approximately 20 cm in depth.

Due to the variability of burrow depths, it is not an easy target to get biodiffusion coefficients to characterize bioturbation for a particular region and species. Therefore, assumptions and approximations must be made for each specific situation.

For earthworms, one approach could be to assume an average burrow depth for each ecological group. Even though anecic species, also known as deep-burrowers, build large, vertical, permanent burrows that may extent 1-2 m deep (Bohlen, 2004; Kladivko), such depths cannot be taken as the characteristic path length for this group. The same applies for all other

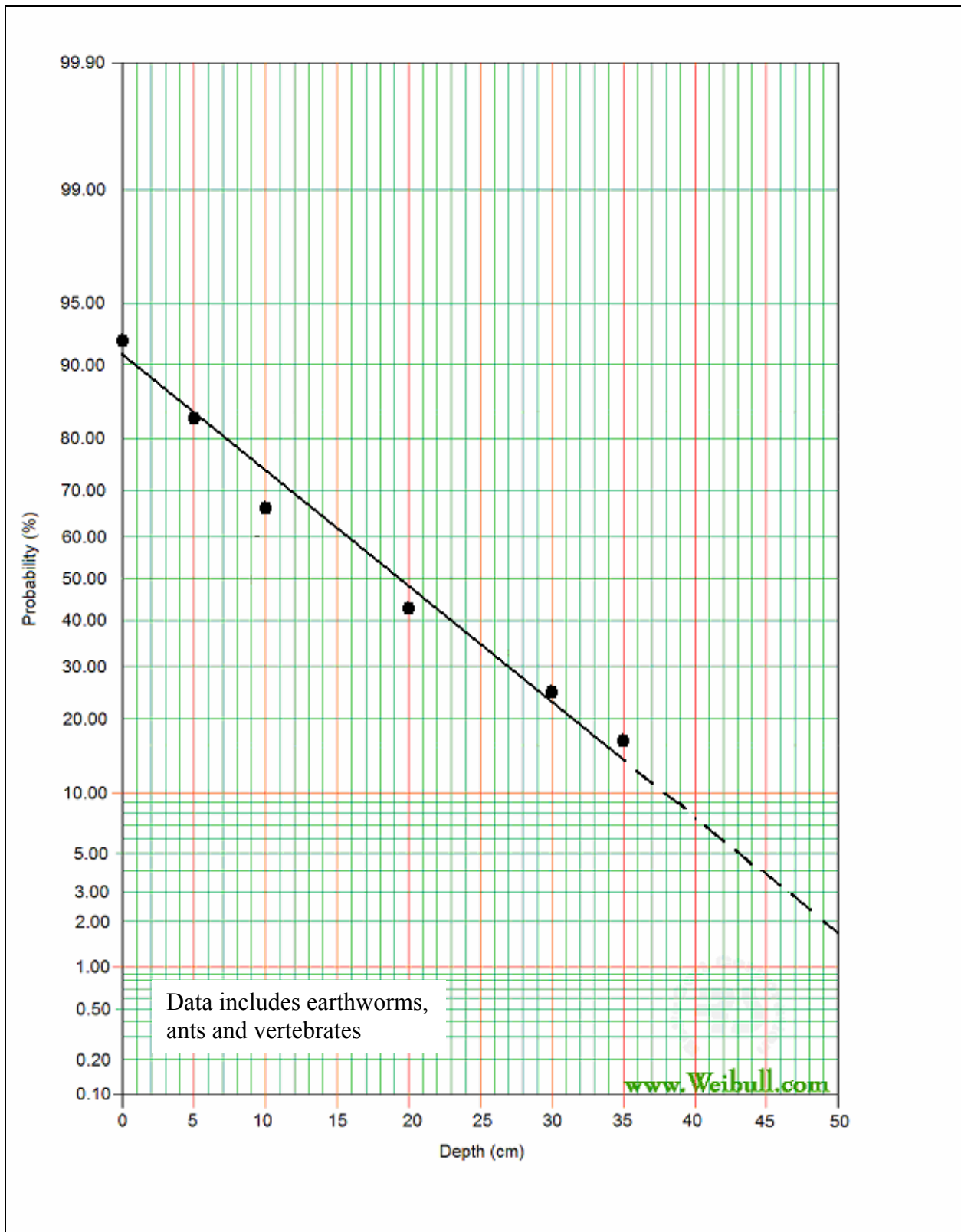


Figure 6. Cumulative probability of definite depth occurrences

earthworm species. Therefore let's assume conservative, average depths based on the information provided within this chapter.

For epigeic earthworms 2 cm will be the assumed burrow depth, 8 cm for endogeic, and 20 cm for anecic. Since lumbricids include species from different groups, mainly endogeic and anecic, a depth of 10 cm will be assumed for this functional group. According to Figure 6 there is 50% or more probability of reaching these depths.

Similarly, characteristic depths need to be assumed for other species. Using the data plotted in Figures 5 and 6, depths in Table 8 will be used later to calculate the corresponding biodiffusion coefficients.

Table 8. Definite depths for different species

<i>Species</i>	<i>Definite Depth (cm)</i>
Epigeic earthworms	2
Endogeic earthworms	8
Anecic earthworms	20
Ants	25
Termites	25
Badgers	20
Gophers	22
Squirrels	12
Kangaroos	10

2.8.2 The Biodiffusion Coefficient

Using Equation 4, soil turnover rates from Table 6, and depths in Table 8, biodiffusion coefficients were computed. The results appear in Tables 9. The ranges shown reflect the ranges in the soil turnover rates.

Table 9. Biodiffusion coefficients for earthworms

<i>Habitat</i>	<i>Location</i>	<i>Earthworm Species</i>	<i>Biodiffusion Coefficient</i>	
			cm ² /year	m ² /day
Tallgrass prairie	Kansas, USA	Diplocardia	0.12	3.3 × 10 ⁻⁸
		Lumbricidae	0.02 - 0.06	0.5 - 1.6 × 10 ⁻⁸
Pasture	France	Lumbricidae	5	1.4 × 10 ⁻⁶
	Australia	A. caliginosa	0.16	4.4 × 10 ⁻⁸
	New Zealand	-	2 - 3	5.5 - 8.2 × 10 ⁻⁷
Forest	Germany	Lumbricidae	4 - 5	1.1 - 1.4 × 10 ⁻⁶
		Megascolecidae	6 - 16	1.6 - 4.4 × 10 ⁻⁶
Shrub savanna	Ivory Coast	Eudrilidae	3.2	8.8 × 10 ⁻⁷
		Microchaetus	4	1.1 × 10 ⁻⁶
Grassland	South Africa	Microchaetus	4	1.1 × 10 ⁻⁶
Pot studies	-	A. caliginosa	1.6 - 2.4	4.4 - 6.6 × 10 ⁻⁷
Poor pasture	England	-	3	8.2 × 10 ⁻⁷
Temperate pastures and grassland	-	-	3 - 4	0.8 - 1.1 × 10 ⁻⁶
Good pasture, no ploughed	England	-	5.6	1.5 × 10 ⁻⁶
Stony field	England	-	2.1	5.8 × 10 ⁻⁷
Swampy field	England	-	4.8 - 5.3	1.3 - 1.5 × 10 ⁻⁶
Dry sandy grassland	England	-	5.6	1.5 × 10 ⁻⁶
Pasture, partially planted	England	-	5.1 - 5.6	1.4 - 1.5 × 10 ⁻⁶

A total of twenty five (n = 25) observations of D₃, including end members of the ranges, are available representing earthworms of various types. The dominant soil type is un-forested prairie, pasture, field and grassland. Only two of the twenty five represent forest lands. All the data appears graphically in Figure 7 on normal cumulative probability paper. There appears to be one spurious data; the value of 16 cm²/year does not fall upon what appears to be a nearly linear relationship (i.e. straight line) suggesting the earthworm data is normally distributed. The average D₃ is 3.35 cm²/year with standard deviation 1.91 cm²/year, excluding the spurious data.

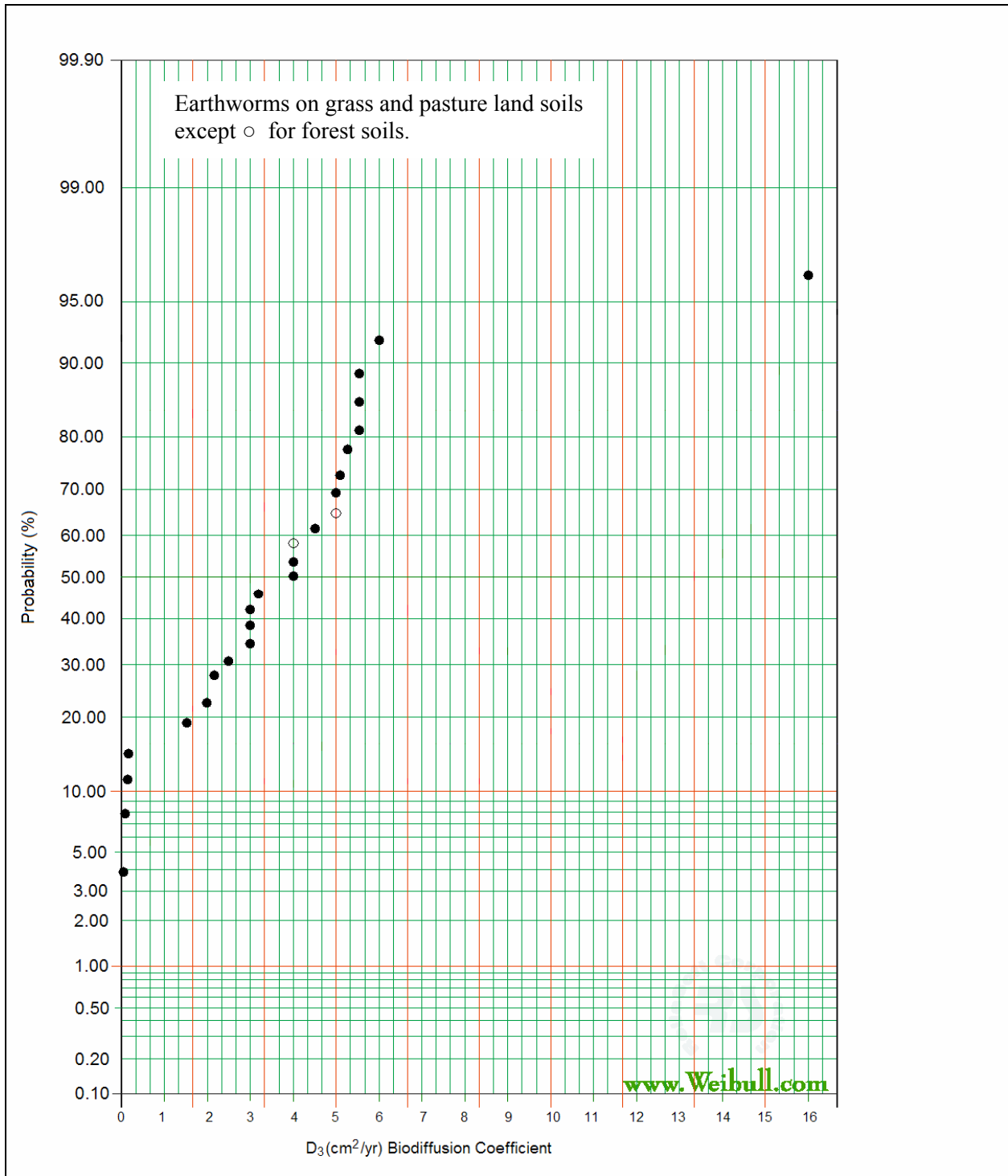


Figure 7. Cumulative probability distribution of earthworm biodiffusion coefficients

Table 10 contains biodiffusion coefficients for ants, termites, and vertebrates, calculated using Equation 4. The turnover rates (v) appear in Table 7 and average depths (h_3) are given in Table 8. This set of biodiffusion coefficients represents some radically different biota types. Three subcategories were created: 1. Ants ($n = 9$), 2. Termites ($n = 7$) and 3. Badgers, gophers, squirrels and kangaroos ($n = 9$). A total of twenty five observations represent the non-earthworm category. An attempt to statistically represent these three data sets on normal probability paper failed with results of very non-linear cumulative probability (CP) correlations. Figure 8 is CP vs log-normal D_3 representation of the three data sets and clearly linear correlations resulted. Three separate sets appear in the same figure for comparison purposes. All three appear to follow slightly different trend lines. The lines shown do not represent statistical fits to the data but only adjusted by eye as an aid in representing the data. Based on this log-linear behavior the average log D_3 's and log σ 's were obtained and converted to the normal numerical values which appear in Table 11.

Mechanical-turbation diffusion coefficients were also calculated, using Equation 6, and plotted in Figure 9. No data exist to represent plowing "biodiffusion" coefficients. The representation in Figure 9 is hypothetical, being parameterized by the plow depth and the frequency of plowing (see Equation 6). However, data was found that characterizes the plow types and depths; these are illustrated in the figure by vertical dashed lines. Accordingly, very different numerical values result from the three primary plow types. Disk harrows used twice yearly have equivalent D_3 values of $\sim 30 \text{ cm}^2/\text{yr}$, moldboard plows $\sim 100 \text{ cm}^2/\text{yr}$ and chisel plows $\sim 1000 \text{ cm}^2/\text{yr}$. Since plowing is done few times per year, these diffusion coefficients are appropriate to model tillage for periods of time of 10 years or more, this being the appropriate

Table 10. Biodiffusion coefficients for different soil biota species

Species	Location	Habitat	Biodiffusion Coefficient	
			cm ² /year	m ² /day
Ants				
	Brazil	<i>not specified</i>	7.75	2.1×10^{-6}
funnel ants	New South Wales, Australia	Semiarid Aeolian soil	0.75	2.1×10^{-7}
	<i>not specified</i>	Semiarid clay soil	0.08	2.1×10^{-8}
mixed species	Kellerberrin, Australia	Rural environments	0.08	2.1×10^{-8}
		Naturally vegetated environments	0.05	1.4×10^{-8}
<i>lasius niger</i>	Michigan, USA	Rural environments	0.18	4.8×10^{-8}
<i>formica exsect.</i>	Wisconsin, USA	Rural environments	2.25	6.2×10^{-7}
<i>formica fusca</i>	USA	Naturally vegetated environments	13.5	3.7×10^{-6}
4 ant species	Chihuahuan Desert, USA	Naturally vegetated environments	0.02	4.8×10^{-9}
Termites				
	<i>not specified</i>	Semiarid environments	0.33 – 5	$0.089 - 1.4 \times 10^{-6}$
	Senegal	Tropical ecosystems	0.35	9.6×10^{-8}
	Uganda	Tropical ecosystems	0.25	6.9×10^{-8}
<i>macrotermes</i> spp.	<i>not specified</i>	Oxisol, Ultisol soil	2.25	6.2×10^{-7}
	Kenya	<i>not specified</i>	0.2	5.5×10^{-8}
<i>macrotermes</i> spp.	Africa	Savannas	2	5.5×10^{-7}
Badgers				
american badger	west-central Idaho, USA	Mixed sagebrush vegetation, fine, silty soils	0.8	2.2×10^{-7}
	New Mexico, USA	Chihuahuan Desert	0.4	1.1×10^{-7}
Gophers				
pocket gopher	USA	<i>not specified</i>	0.75 – 13	$0.2 - 3.6 \times 10^{-6}$
pocket gopher	New Mexico, USA	Chihuahuan Desert, grassland community	15.4	4.2×10^{-6}
pocket gopher	USA	<i>not specified</i>	max. 18	max. 4.9×10^{-6}
pocket gopher	<i>not specified</i>	<i>not specified</i>	3.3	9×10^{-7}
Squirrels				
prairie dog	<i>not specified</i>	Ustoll soil	6.48	1.8×10^{-6}
arctic ground squirrel	<i>not specified</i>	<i>not specified</i>	1.8	4.9×10^{-7}
Kangaroos				
grey kangaroo	western New South Wales, Australia	Semiarid wooded rangeland	0.2	5.5×10^{-8}

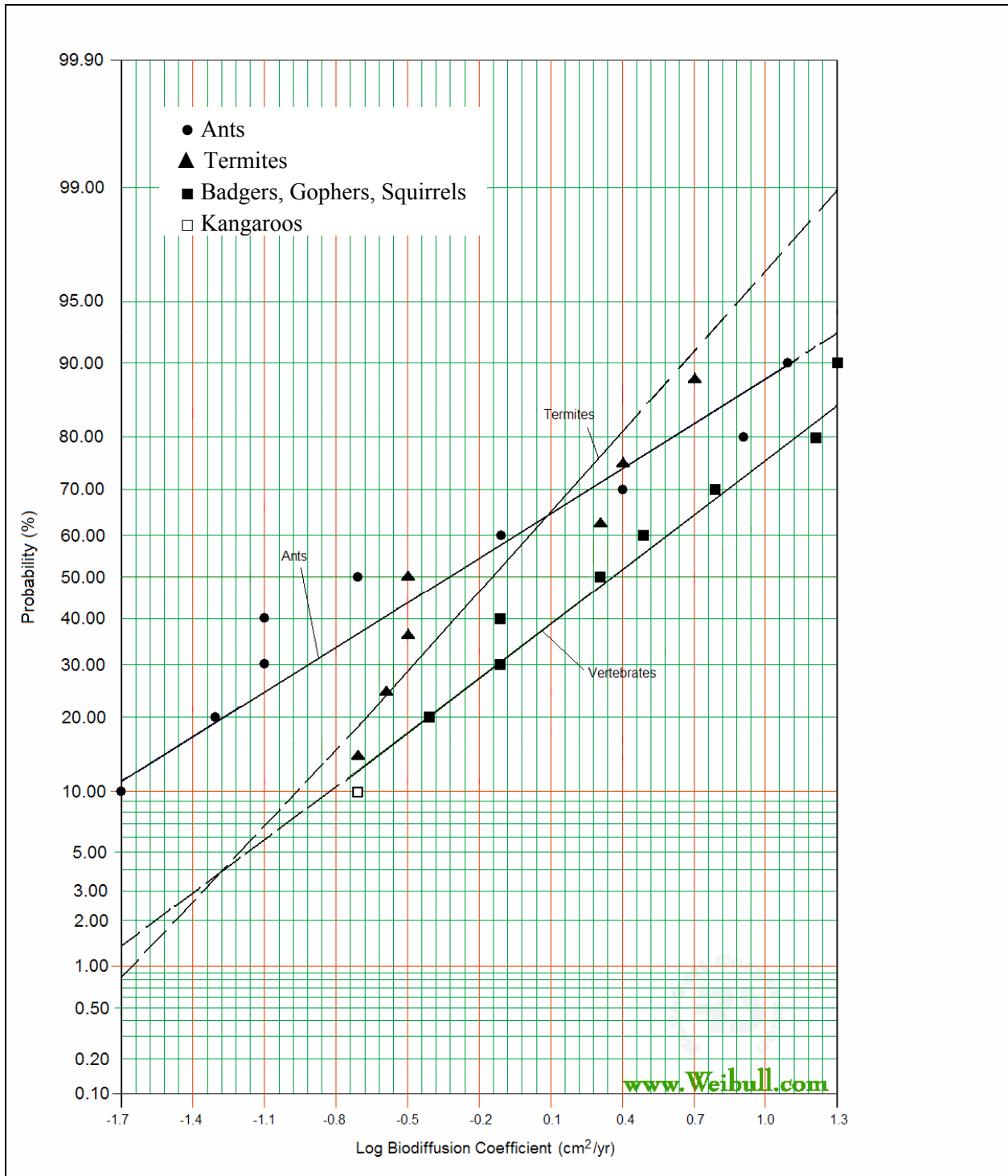


Figure 8. Cumulative probability distribution of different species bioturbation coefficients

Table 11. Biodiffusion coefficient statistics

<i>Biota</i>	<i>n</i>	$D_3 - \sigma$	D_3 (cm^2/yr)	$D_3 + \sigma$
Earthworms	25	1.44	3.35	5.26
Ants	9	0.043	0.389	3.52
Termites	7	0.226	0.747	2.46
B, G, S, K*	9	0.445	2.00	9.00

*Badgers, Gophers, Squirrels and Kangaroos

time-scale needed for long-term steady state chemical models. Otherwise transient chemical models are needed to capture the turnover events.

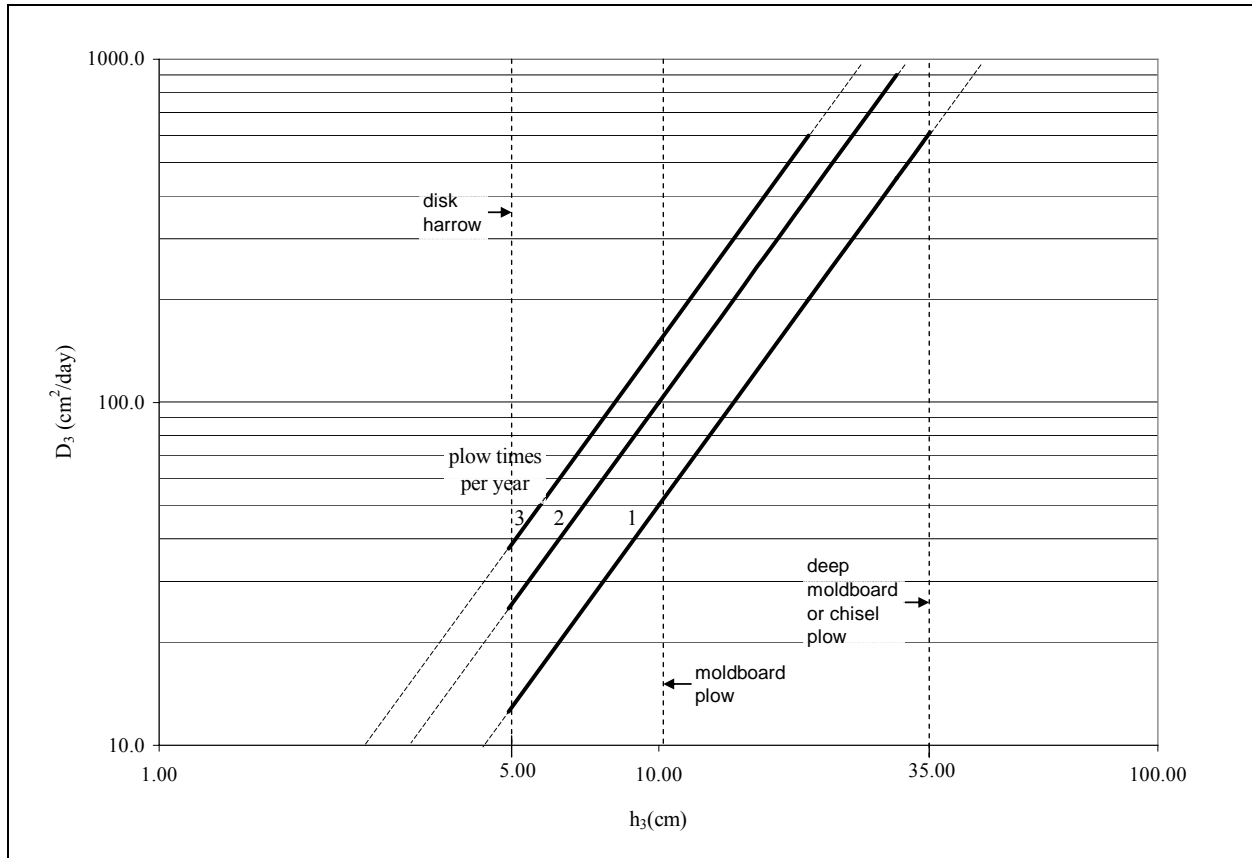


Figure 9. Tillage practices modeling

2.8.3 Geographic Distribution

As previously mentioned several are the factors that influence bioturbation, as well as the distribution of its responsible species. Soil biodiversity tends to be greater in forests compared to grasslands and in undisturbed natural lands compared to cultivated fields. However the number and type of organisms vary from one geographic/temperate system and environment to another.

Based on the data presented in the preceding tables and information published by several authors (DesertUSA.com; SWCS, 2000; Whitford & Kay, 1999; Eldridge et al., 2002; Eldridge, 2004; Lal, 1988; de Bruyn, 1999) a general habitat distribution for the species studied is summarized in Table 12. Although limited in number the species include several geographic/temperate ecoregions of the earth. These are: tropical, temperate, semi-arid and the arid region.

Table 12. General species distribution

<i>Species</i>	<i>Predominant Habitat</i>
Earthworms	Temperate and tropical soils, mainly in humid and sub-humid tropics. More abundant in grasslands and temperate woodlands than in croplands.
Ants and termites	More active in arid and semi-arid regions. Higher densities in the Tropics and Australia.
Pocket gophers	From the deserts up into the mountains, mainly arid and semi-arid regions. All of Arizona, southern half of Utah, most of California, western half of New Mexico, and southern third of Nevada.
Badgers	Deserts, grasslands, regular inhabitants of shrub-steppe ecosystems. Predominant in western United States.
Kangaroos	Common in woodlands and shrublands in arid and semi-arid eastern Australia.

Bailey's ecoregion classification was used to make the USA map appearing in Figure 10. It is for illustrative purposes. The broadest classification level is the domain, which contain groups of related climates, differentiated based on precipitation and temperature. Two prevail in the USA, Humid Temperate and Dry. Divisions within the domains are differentiated based on precipitation levels and patterns as well as temperature. These divisions are subdivided into provinces, shown in Figure 10. Provinces are differentiated based on vegetation or other natural land covers. This map was created using ArcGIS software and layers from National Atlas of United States. Detailed description of climate, vegetation, soils, and fauna, for each province can be found at the National Atlas of United States website (see references).

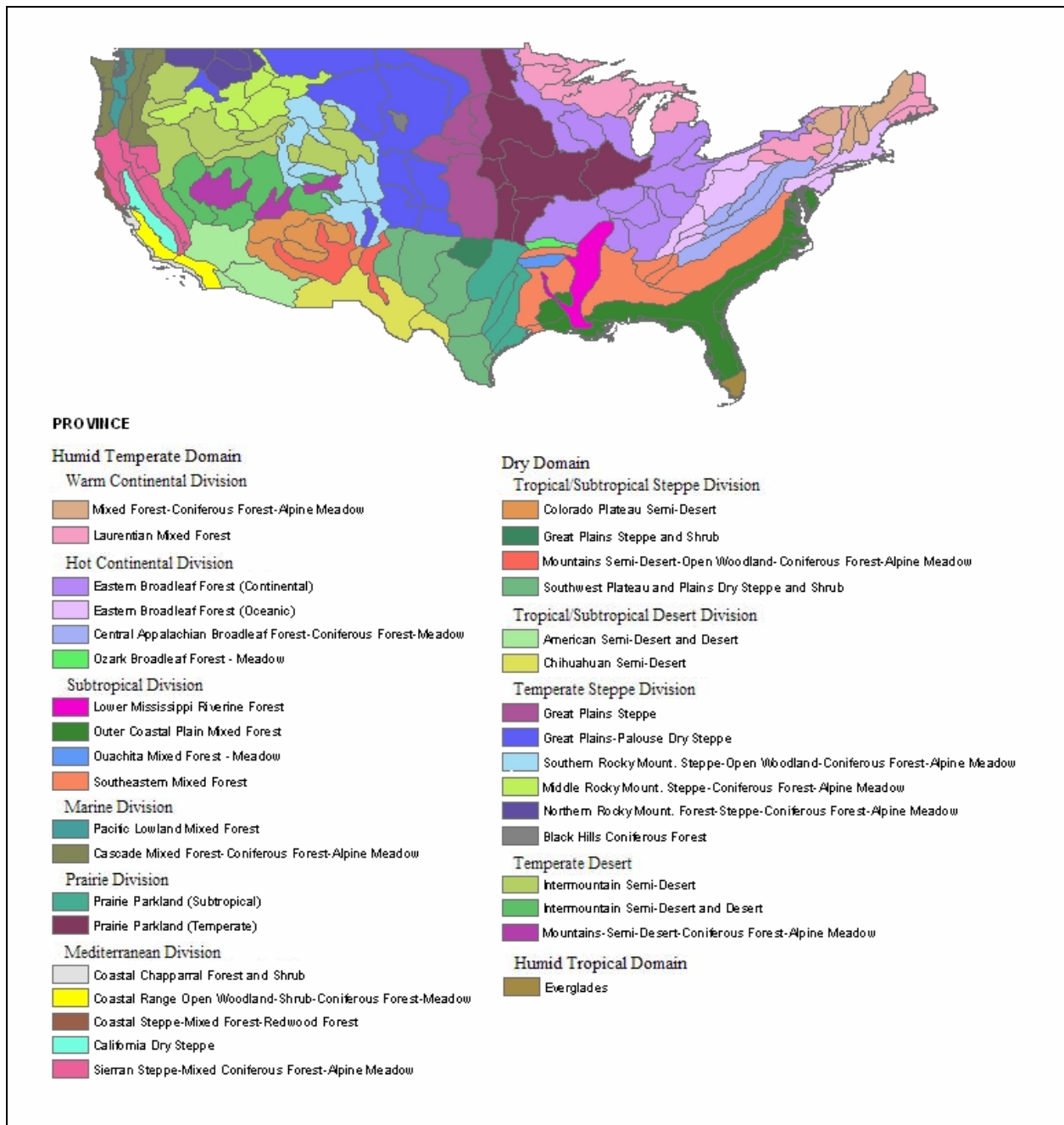


Figure 10. Bailey's Ecoregions of the United States

Ideally, such maps can be constructed for countries, sub-continent and continents as an aid in classifying the types of species likely to dominate bioturbation including plowing by humans. This together with biodiffusion coefficient magnitudes and area attribution can be used to estimate characteristic average biodiffusion coefficient values for defined ecoregions.

According to the general species distribution described in Table 11 earthworms are more likely to be found in the humid temperate and tropical domains of the USA and ants and pocket gophers in the dry domain.

2.8.4 Discussion of Results

It can be seen from the information provided in this chapter that there are many contributors to bioturbation, even though not all of them were described. Additionally, since each of these species bioturbate in different ways and depending on several environmental conditions, it is not easy to catalog and rank their importance. However, this is necessary in order to find the right parameters to model bioturbation in specific situations.

It is important to select some parameters to rank the species considered. According to the purpose of the present study this ranking will be done based on soil turnover rates. The turnover rates presented in Tables 6 and 7 were used to make Figure 11, which shows maximum and minimum rates by each taxon.

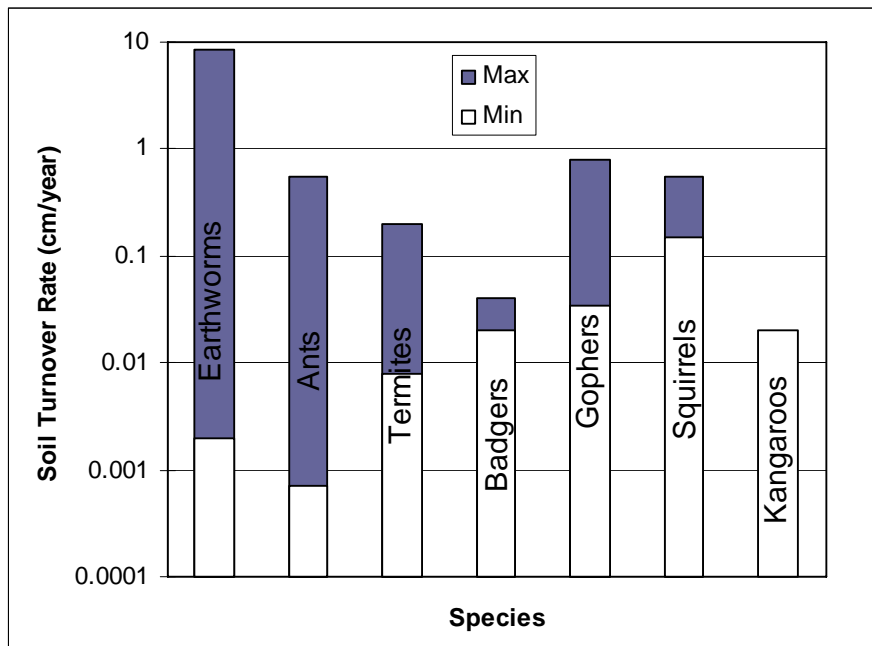


Figure 11. Maximum and minimum soil turnover rates by different species

The results shown in the previous chart agree with what have been reported about earthworms by several authors; they are the most significant contributors to bioturbation (McLachlan et al., 2002). Equivalent results would be found if using bioturbation diffusion coefficients from Tables 9 and 10 instead of soil turnover rates, as shown in Figure 12.

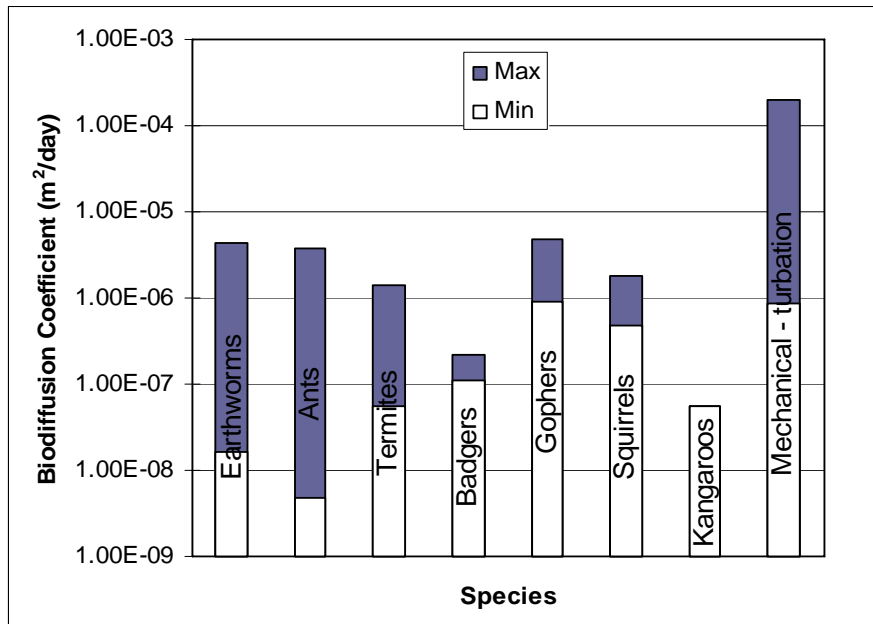


Figure 12. Maximum and minimum biodiffusion coefficients

Based on the organic matter transport and degradation rates assigned to earthworms, the significance of bioturbation by these invertebrates is confirmed, and a more explicit consideration of their activity seems warranted (Armitage et al., unpublished). According to a study done by Humphreys and Field (1998) in a site close to Sydney (Australia), the mixing fauna was dominated by earthworms and ants. Therefore, in general, earthworms soil turnover rates and depths reached by them can be used to characterize bioturbation.

In Figure 12 the magnitude of the biodiffusion coefficients is not as variable among taxa as the observed with soil turnover rates. The biodiffusion coefficient interval for mechanical-turbation corresponding to tillage done once a year is also included in Figure 12, and according with its magnitude mixing of soil by this process is significantly high compared to bioturbation.

However, it must be recalled that this is a periodic rather than a continuous process and the coefficient is meaningful when considering tillage for a long period of time. On the other hand, since bioturbation represents constant mixing, data for less than a year can be modeled with the bioturbation diffusion coefficients reported.

There is clear evidence from numerous studies that several species of surface dwelling invertebrates and vertebrates occupy the soil surface. In doing so and as part of their life-cycle activities they move soil particles both vertically and laterally. The maximum depth of penetration of these activities is definitely known to be 35 cm with some few observed at the 100 cm depth. A 20 cm depth represents the average of all data reported. However, 10 to 12 cm may be a more realistic average for earthworms, the most common and ubiquitous organism found at the soil surface.

The biodiffusion coefficient is calculated parameter based on soil turnover rates and the average bioturbed depth which are both observed (i.e. measured) quantities. Although the solid particles do not “diffuse” across regions in the soil with high and low particle concentration the term reflects mixing and re-distribution consistent with Gaussian characteristics. The latter also mimics the chemical transport process in gas, liquid and solid media which are known to be diffusion processes. Therefore the biodiffusion coefficient is defined so that it may be used in diffusion transport chemical models to track chemical species known to adsorb to the soil particles (see Section 3).

A total of fifty observations of soil surface biodiffusion coefficients were obtained; twenty five represent earthworms. The earthworm D_3 values appear to be normally distributed about a mean of 3.4 cm²/year with a relative small standard deviation being 36%. All other species biodiffusion data follow a log-normal distribution fairly well. It was used to obtain the

average and deviations shown in Table 11. As a group of bioturbators badgers, gophers, squirrels and kangaroos (BGSK) displayed an average D_3 of 2.00 cm²/year. Termites ranked next with 0.75 cm²/year and ants' value was 0.39 cm²/year. Unlike the earthworm data the D_3 values for the BGSK group, termites and ants varied greatly in the low to high values. As shown in Table 11 the high to low ratio was 3.7 for earthworms and 20, 11, and 82 for BGSK group, termites, and ants, respectively.

It can only be speculated as to the reasons earthworms and the three other biota types display somewhat different D_3 characteristics. The earthworm D_3 data follows a normal distribution with relative low variability whereas the others follow log-normal distributions of D_3 values and display a large range of variation about the means. The organisms in the BGSK group have legs as well as do ants and termites. The appendages as well as the bodies of the organisms can play a role in moving soil particles. The legs being responsible for scattering about small size particles and the bodies shoving the larger pieces. This may explain the range of D_3 values observed. Earthworms have no appendages plus the process of particle movement is dominated by ingestion and defecation. The latter process limits the size of particles moved to a specific range of values. These behavior factors may limit the magnitude of the D_3 values observed for earthworms.

The above differences in behavior provide no basis for the differences in magnitudes of the D_3 for each biota type. Although considering the range of values the termites and ants data may constitute a single set. The small sizes of the data sets make further analysis with respect to D_3 magnitudes problematic.

3. THEORETICAL CHEMICAL MODELS

Risk-based environmental decision making is crucial to effectively respond to environmental problems. In order to give an appropriate response, quantitative tools are necessary. These include conceptual, physical, and mathematical models of the environmental and receptor processes that transport, dilute, degrade, and mitigate the effects of environmental contaminants. Chemical fate and transport (CFaT) models must represent a good understanding and description of these key processes responsible for the movement of chemicals. However, due to their complexity in the environment, it is not possible to develop models that quantitatively describe all processes and all their effects (Choy et al., 1999). Soil is one of the most important compartments where such processes continuously take place.

The following is a model developed to make predictions of the behavior of organic chemicals in soils. Some assumptions have to be made because of the complexity of the fate processes and the compartment being studied. This model treats the soil as a homogeneous box with uniform properties, such as density, temperature, organic carbon content, and porosity. However it is known that natural soils are not uniform and these properties vary over the depths considered.

The only transport process included in the model is diffusion through all soil phases: air-, water-, and solid-phase. For chemicals with low solubility in water, e.g. polychlorinated biphenyls, it is valid to neglect advection and convection, assuming that there is no water flux within the soil. Degradation of the chemical is also neglected, and chemical concentration is assumed to be homogeneous in the horizontal plane. For these considerations, the continuity equation for a chemical A moving in the vertical direction (Z), as shown in Figure 13, reduces to

$$\frac{\partial \rho_{A3}}{\partial t} + \frac{\partial n_{Az}}{\partial z} = 0 \quad (7)$$

where: $\rho_{A3} \equiv$ total concentration of A in the soil

$n_{Az} \equiv$ flux of chemical A in the Z direction

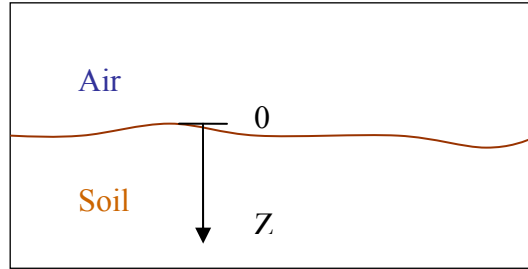


Figure 13. Soil layers at air-soil interface

The flux is given by

$$n_{Az} = -D_{A1} \frac{d\rho_{A1}}{dz} - D_{A2} \frac{d\rho_{A2}}{dz} - \rho_b D_{AS} \frac{d\omega_A}{dz} \quad (8)$$

where: $\rho_{A1} \equiv$ concentration of A in the air-phase

$\rho_{A2} \equiv$ concentration of A in the water-phase

$\omega_A \equiv$ concentration of A in the solid-phase

$D_{A1} \equiv$ diffusion coefficient of A in pores filled with air

$D_{A2} \equiv$ diffusion coefficient of A in pores filled with water

$D_{AS} \equiv$ diffusion coefficient of A sorbed to soil particles

Distribution or partition coefficients are necessary to describe how chemicals are distributed between two phases. The fate and transport of chemicals in the environment is often limited to consideration of the equilibrium partitioning of these contaminants between environmental phases (Choy et al., 1999). Although concentrations in soil and other phases in natural systems frequently deviates from those at equilibrium, the equilibrium data serve as an essential guide to describe contaminant movement at a particular point in time (Chiou, 2002). Therefore, equilibrium between phases will be assumed in this model even though the entire soil

is seldom in equilibrium, because it alternately wet and dries, swells and shrinks, hardens and softens, freezes and thaws, compacts and cracks, adsorbs and emits gases (Hillel, 2004).

Henry's law can be used to express air-water equilibrium, being Henry's constant the distribution coefficient,

$$\rho_{A1} = H_p \cdot \rho_{A2} \quad (9)$$

where: $H_p \equiv$ Henry's constant

The solid phase in the soil interacts with the fluids, water and air, that adjoin it in the soil pores (Hillel, 2004). Equilibrium partitioning between solid and water phases can be expressed as

$$\omega_A = K_{A32}^* \cdot \rho_{A2} \quad (10)$$

where: $K_{A32}^* \equiv$ soil-water partition coefficient

The soil solid phase is formed by organic and mineral fractions. A simplified common method of accounting for sorption of organic chemicals in soils is based on the assumption that organics attach to the natural organic matter in the soil. It has been also suggested that solubility in the organic matter is an appropriate mechanism to explain soil-water distribution of non-ionic organic compounds (Thibodeaux, 1996). Therefore, the soil-water partition coefficient can be approximated to

$$K_{A32}^* = k_{OC} \cdot f_{OC} \quad (11)$$

where: $k_{OC} \equiv$ organic carbon partition coefficient

$f_{OC} \equiv$ fraction of organic carbon in the soil

Consequently, concentration of chemical in the solid phase can be written as

$$\omega_A = k_{OC} \cdot f_{OC} \cdot \rho_{A2} \quad (12)$$

Assuming chemical equilibrium between soil phases at all times, as given above, Equation 8 can be rewritten as follows

$$n_{Az} = - \left(\frac{D_{A1} H_{\rho}}{K_{A32}^*} + \frac{D_{A2}}{K_{A32}^*} + \rho_b D_{AS} \right) \frac{d\omega_A}{dz} \quad (13)$$

Air and water phases in the soil are present filling soil pores, in proportions that vary with soil type, depth, and pores characteristics, among others. Since chemical A is distributed among all three phases, its total concentration in the soil can be obtained adding up three fractions as follows

$$\rho_{A3} = \varepsilon_1 \cdot \rho_{A1} + \varepsilon_2 \cdot \rho_{A2} + \rho_b \cdot \omega_A \quad (14)$$

where: $\varepsilon_1 \equiv$ pore fraction filled with air

$\varepsilon_2 \equiv$ pore fraction filled with water

Diffusion coefficients for air and water are calculated using Millington-Quirk model, multiplying molecular diffusion coefficients in each phase by a porosity factor, the latter to account for the reduced flow area and the increased path length for molecules of A diffusing in the soil (Thibodeaux, 1996).

$$D_{A1} = \left(\frac{\varepsilon_1^{10/3}}{\varepsilon^2} \right) \cdot D_{A1m} \quad (15)$$

$$D_{A2} = \left(\frac{\varepsilon_2^{10/3}}{\varepsilon^2} \right) \cdot D_{A2m} \quad (16)$$

where: $\varepsilon \equiv$ total soil porosity

$D_{A1m} \equiv$ molecular diffusion coefficient in air

$D_{A2m} \equiv$ molecular diffusion coefficient in water

Using the given expressions for equilibrium between phases, Equations 9 and 10, ρ_{A1} and ρ_{A2} can be replaced in Equation 14 giving

$$\rho_{A3} = \omega_A \left(\frac{\varepsilon_1 H_\rho}{K_{A32}^*} + \frac{\varepsilon_2}{K_{A32}^*} + \rho_b \right) \quad (17)$$

Replacing Equation 13 and 17 into Equation 7 gives

$$\left(\frac{\varepsilon_1 H_\rho}{K_{A32}^*} + \frac{\varepsilon_2}{K_{A32}^*} + \rho_b \right) \frac{\partial \omega_A}{\partial t} - \left(\frac{D_{A1} H_\rho}{K_{A32}^*} + \frac{D_{A2}}{K_{A32}^*} + \rho_b D_{AS} \right) \frac{\partial^2 \omega_A}{\partial z^2} = 0 \quad (18)$$

To simplify the previous expression, let's group both terms in parenthesis and define a new parameter as follows

$$D'_{A3} = \frac{\left(\frac{D_{A1} H_\rho}{K_{A32}^*} + \frac{D_{A2}}{K_{A32}^*} + \rho_b D_{AS} \right)}{\left(\frac{\varepsilon_1 H_\rho}{K_{A32}^*} + \frac{\varepsilon_2}{K_{A32}^*} + \rho_b \right)} \quad (19)$$

The denominator of the latter expression will be defined as a new variable K

$$K = \frac{\varepsilon_1 H_\rho}{K_{A32}^*} + \frac{\varepsilon_2}{K_{A32}^*} + \rho_b \quad (20)$$

Then, an effective overall soil diffusivity (D_{A3}) can be obtained as follows

$$D_{A3} = \frac{D'_{A3} \cdot K}{\rho_b} = \frac{D_{A1} H_\rho}{\rho_b K_{A32}^*} + \frac{D_{A2}}{\rho_b K_{A32}^*} + D_{AS} \quad (21)$$

Rearranging Equation 18 gives

$$\frac{\partial \omega_A}{\partial t} - D'_{A3} \frac{\partial^2 \omega_A}{\partial z^2} = 0 \quad (22)$$

To solve the differential equation above, initial and boundary conditions are needed. At the beginning it is assumed that there is no chemical in the soil. At the air-soil interface a

concentration ω_{Ai} is maintained all the time, and it is also assumed that chemical A is always absent very deep into the soil. Therefore, conditions are:

Initial Condition:

$$\text{at } t = 0, \omega_A = 0 \text{ for all } z$$

Boundary conditions:

$$\text{at } z = 0, \omega_A = \omega_{Ai}$$

$$\text{at } z \rightarrow \infty, \omega_A = 0$$

Solving Equation 22 and evaluating the previous conditions gives the well known result:

$$\omega_A = \omega_{Ai} \cdot \operatorname{erfc} \left(\frac{z}{2\sqrt{(D_{A3}/K) \cdot t}} \right) \quad (23)$$

Concentration of chemicals or contaminants at different depths after a given time can be obtained using Equation 23. Concentration at the air-soil interface can be estimated if the concentration in air is known, assuming chemical equilibrium between bulk environmental phases.

The solid-phase diffusion coefficient D_{AS} , enclosed in the effective diffusion coefficient, is the one related to bioturbation and other processes responsible of soil particles movement. Values for this parameter have to be selected based on the biodiffusion coefficients given in Chapter 2 and the analysis on species and ecoregions presented, or determined from concentration data, if known.

4. QUANTITATIVE METHODS

There are numerous organic wastes from industrial operations or from use and disposal of manufactured products that may contaminate air, land, and water, from effluents, leakage of waste dumps, or accidental spills and fires (Spiro et al., 1996). In the United States, the Environmental Protection Agency (EPA) is the primary regulatory agency that protects the environment from chemical pollutants. Since its establishment in 1970, the understanding of the generation of chemical wastes, their controllability, and their fate and transport in the environment, has improved significantly (Cohen, 1998).

Polychlorinated biphenyls (PCBs) are a group of organic chemicals containing 209 isomers (congeners). The U.S. EPA treats PCBs as being potentially hazardous, probable human carcinogens, and has developed regulations and policies for their management. Even though PCBs are no longer produced or used in the United States, they were already present in soil and water, and are redistributed in the environment (EPA, 2000).

An input of semi-volatile organic compounds (SOCs), such as PCBs and PAHs (polychlorinated aromatic hydrocarbons), is received by most soils from atmospheric deposition. On the other hand, PCBs are environmentally persistent organic pollutants, and bioaccumulate in the food chain (their concentration increases as they move from simple aquatic life forms, to fish, to humans) (EPA, 1997). Also, it has been hypothesized that soils have acted as a significant repository of these chemicals, initially absorbing the substance and then releasing it slowly back to the atmosphere (Backe et al., 2004). For all previous reasons, assessment of the vertical distribution of these organic chlorine compounds in soils is relevant for environmental purposes.

Several mass-balance models that simulate the distribution of organic chemicals in soils and their exchange with the atmosphere have been developed, similar to the one presented in the

preceding chapter. Some of them are fugacity-type models (Harner et al., 1995; Cousins et al., 1999b, Backe et al., 2004), while others are based on concentrations (Jury et al., 1983; McLachlan et al., 2002).

The mathematical model described in Chapter 3 along with field data on soil chemical profiles will be used to represent the distribution of PCBs in soils, in order to extract the bioturbation transport parameter D_{AS} characterizing such distribution.

4.1 PCBs Field Data

Cousins et al. (1999a) collected soil cores at four locations in the UK and analyzed them for PCBs. One of the sites was a grassland park whose soil has two distinct horizons, a brown organic one (0-5 cm depth), and a lower mineral horizon (5-20 cm). Measured concentrations at this site are presented in Table 13, and plotted in Figures 14 and 15. The selected congeners are:

- PCB-52 ($C_{12}H_6Cl_4$): 2,2',5,5'-tetrachlorobiphenyl
- PCB-101 ($C_{12}H_5Cl_5$): 2,2',4,5,5'-pentachlorobiphenyl
- PCB-153 ($C_{12}H_4Cl_6$): 2,2',4,4',5,5': hexachlorobiphenyl

In addition total PCBs concentration was used.

Table 13. Measured concentrations of selected PCB congeners

		Measured concentrations (ng/g)			
		Depth (m)	PCB-52	PCB-153	PCB-101
upper zone	0.005	0.12	0.29	0.14	3.2
	0.015	0.14	0.36	0.14	3
	0.025	0.15	0.22	0.16	4.3
	0.035	0.12	0.16	0.1	3.4
lower zone	0.045	0.08	0.09	0.085	2.9
	0.06	0.05	0.082	0.072	1.6
	0.08	0.045	0.079	0.066	1.3
	0.1	0.033	0.061	0.047	1.5
	0.12	0.022	0.044	0.033	1.2
	0.14	0.009	0.02	0.015	0.58
	0.16	0.014	0.032	0.028	1.1

* includes total concentration of 53 congeners

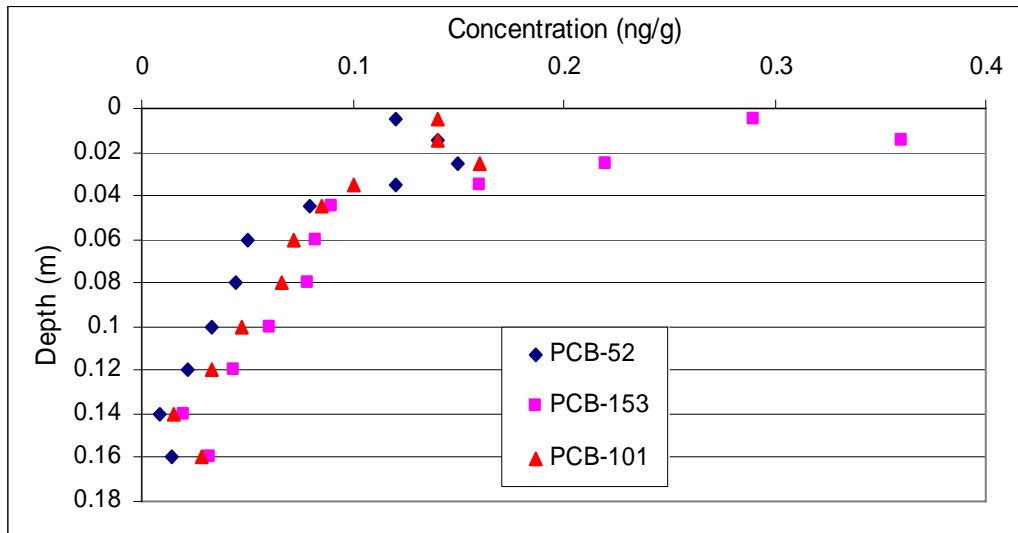


Figure 14. Soil vertical distribution of selected PCBs

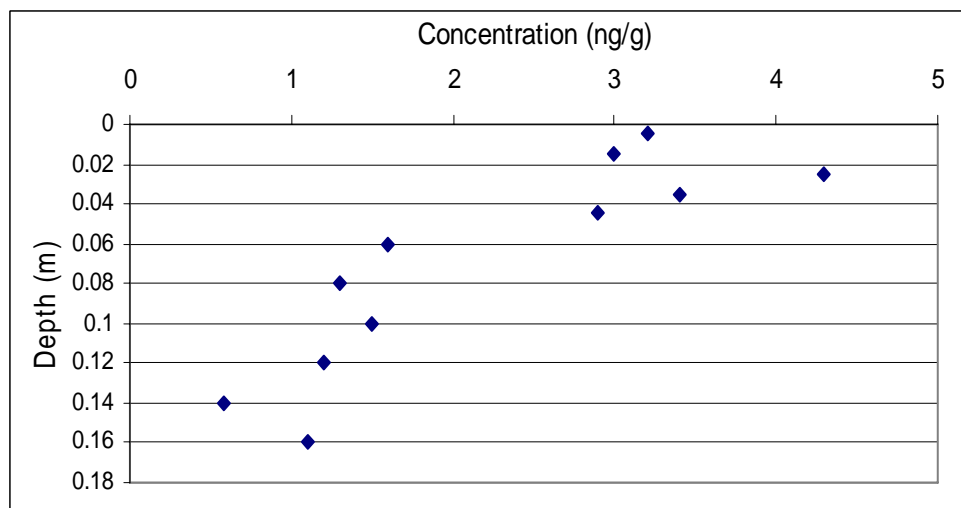


Figure 15. Soil vertical distribution for 53 PCB congeners

From the plots it can be seen that the concentrations decrease with increasing depth suggesting input at the air-soil interface.

Properties of these PCBs, such as Henry's constant, partition coefficients, and molecular diffusivities, are needed to calculate D'_{A3} (Eq. 19) and determine the contribution of each phase to the fate of chemicals in soils, which is an important contribution of the model. These properties are presented in Table 14. Molecular diffusivities were obtained from Mackay et al. (1992), same used by Cousins et al. (1999b)

Table 14. Properties of selected PCB congeners

	PCB-52	PCB-153	PCB-101
H ρ *	9.12E-03	6.61E-03	7.76E-03
Log koc**	4.65	5.09	4.87
koc (L/kg)	4.48E+04	1.23E+05	7.41E+04
D _{A1m} (m ² /d)	0.43	0.43	0.43
D _{A2m} (m ² /d)	4.30E-05	4.30E-05	4.30E-05
*using group contribution method from EPA's EPISUITE			
**using EPA's EPISUITE			

Soil parameters are also needed for further calculations. As already mentioned, properties of natural soils vary with depth, however for this study a soil with uniform properties will be assumed. Under optimal conditions for growth of plants, liquid and gas components of soil constitute about 50%, 25% each, while solids occupy the other 50%. The inorganic fraction of total solids comprises more than 95% by weight for most mineral soils (Lal et al., 2004). Organic solids form a small fraction but play an important role in numerous soil processes. The soil properties assumed for this model are shown in Table 15 and are an average of the properties measured by Cousins et al (1999b) in the grassland park soil core. Similar soil properties were employed by Jury et al. (1983) and McLachlan et al. (2002). Also, these are within the range of soil physical properties presented by Lal et al. (2004) in relation to plant growth.

Table 15. Soil properties

ε_1	0.2
ε_2	0.3
ε	0.5
ρ_b (kg/m ³)	1350
f_{oc}	0.034

Modeling of the vertical distribution data on Figures 14 and 15 will be done using Equation 23, assuming the unknown parameters that give the smallest error. A trial and error procedure was followed for each PCB in this manner:

- Get a first approximated value for the concentration at the interface (ω_{Ai}) making a crude extrapolation of the PCB concentration to the surface
- Assume a value for the model adjustable parameter D'_{A3}
- Assume 20 years as the time period for deposition of PCB from air. This time is unknown, but we recognize it is in the order of several decades, when PCB use began.
- Replace values in Equation 23, and get concentrations for all depths
- Calculate a statistical least squares error, as follows

$$e^2 = \sum_{j=1}^{11} (\omega_{AM} - \omega_{AD})_j^2 \quad (24)$$

where: $\omega_{AM} \equiv$ model estimated concentration based on assumed ω_{Ai} , D_{A3}/K , and t , at z

$\omega_{AD} \equiv$ concentration data (given in Table 13) at the same z

- Repeat all steps until getting the smallest error.

4.2 Modeling Results

Table 16 contains ω_{Ai} and D_{A3}/K assumed that give the smallest errors, and Table 17 presents concentrations and errors calculated following the previous steps. For PCB-153 the depth was divided in two zones, upper and lower, to get a better fit.

Table 16. Fitted parameters obtained for the model

Chemical	ω_{A3i} (ng/g)	D'_{A3} (m ² /d)	D'_{A3} (cm ² /yr)
PCB-52	0.156	4.40E-07	1.61
PCB-153 (upper)	0.370	1.80E-07	0.66
PCB-153 (lower)	0.160	7.50E-07	2.74
PCB-101	0.161	6.00E-07	2.19
PCBs	4.300	6.70E-07	2.45

Alternative values for D'_{A3} shown in Table 16 represent a deposition time of 20 years, which was assumed. As shown in Equation 23, the product $D'_{A3} \cdot t$ is the total adjustable fitting parameter, since t is arbitrarily chosen. If an alternative time t is assumed then an alternative

D'_{A3} can be obtained. For example if 40 years is used, the D'_{A3} for PCB-52 becomes $2.2E-07$ m^2/day . The point is that even the time is twice as long, the D'_{A3} ratio is halved.

Table 17. Model calculated concentrations of selected PCB congeners

Depth (m)	Calculated Concentration (ng/g)				e^2			
	PCB-52	PCB-153	PCB-101	PCBs	PCB-52	PCB-153	PCB-101	PCBs
0.005	0.148	0.341	0.154	4.127	7.98E-04	2.63E-03	2.00E-04	0.859
0.015	0.133	0.285	0.141	3.782	5.13E-05	5.65E-03	2.50E-07	0.611
0.025	0.118	0.232	0.127	3.442	1.04E-03	1.33E-04	1.08E-03	0.736
0.035	0.103	0.183	0.114	3.111	2.78E-04	5.32E-04	1.98E-04	0.084
0.045	0.090	0.107	0.102	2.791	9.26E-05	2.80E-04	2.73E-04	0.012
0.06	0.071	0.091	0.084	2.340	4.34E-04	7.43E-05	1.43E-04	0.547
0.08	0.050	0.071	0.063	1.800	2.15E-05	6.19E-05	7.71E-06	0.250
0.1	0.033	0.054	0.046	1.342	9.24E-09	4.51E-05	1.13E-06	0.025
0.12	0.021	0.040	0.032	0.968	1.09E-06	1.42E-05	6.93E-07	0.054
0.14	0.013	0.029	0.022	0.675	1.29E-05	8.01E-05	4.47E-05	0.009
0.16	0.007	0.020	0.014	0.455	4.68E-05	1.39E-04	1.94E-04	0.417
				SUM	0.0028	0.0096	0.0021	3.603

Concentrations in Table 17 are plotted in Figures 16, 17, 18, and 19, together with the field data published by Cousins et al. (1999b) presented in Table 13, to visualize the fits obtained with the model.

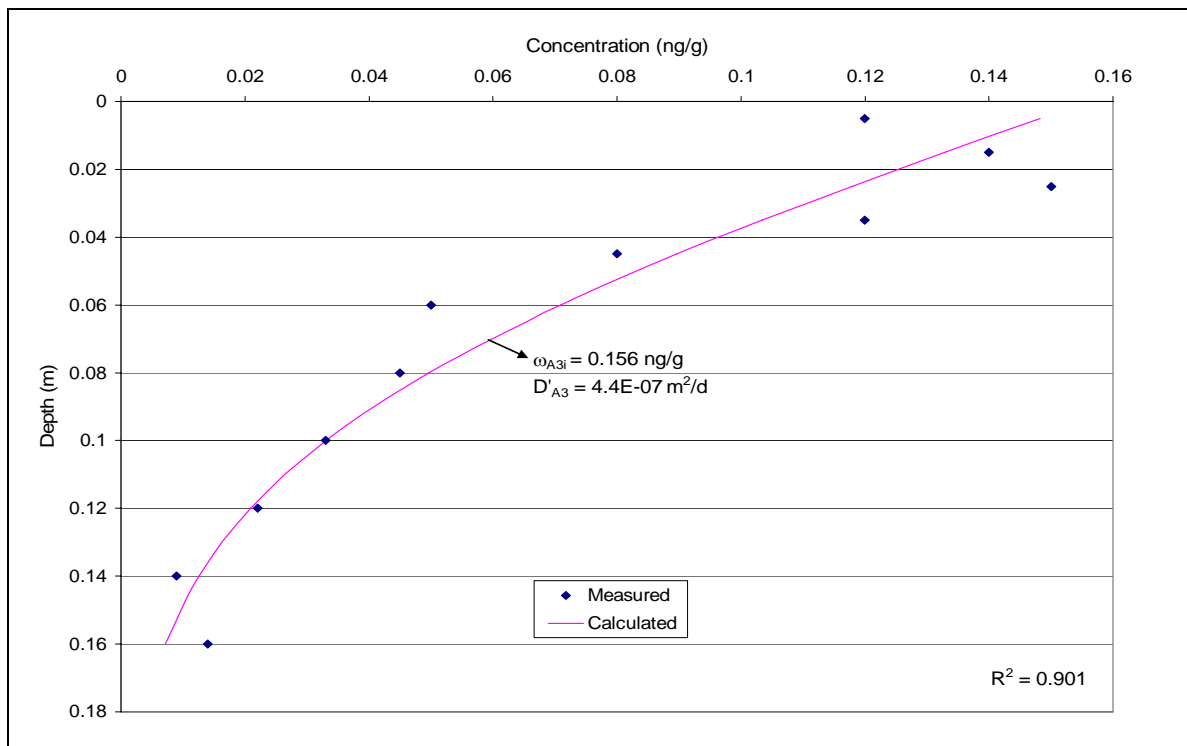


Figure 16. Vertical predicted distribution of PCB-52 in soil

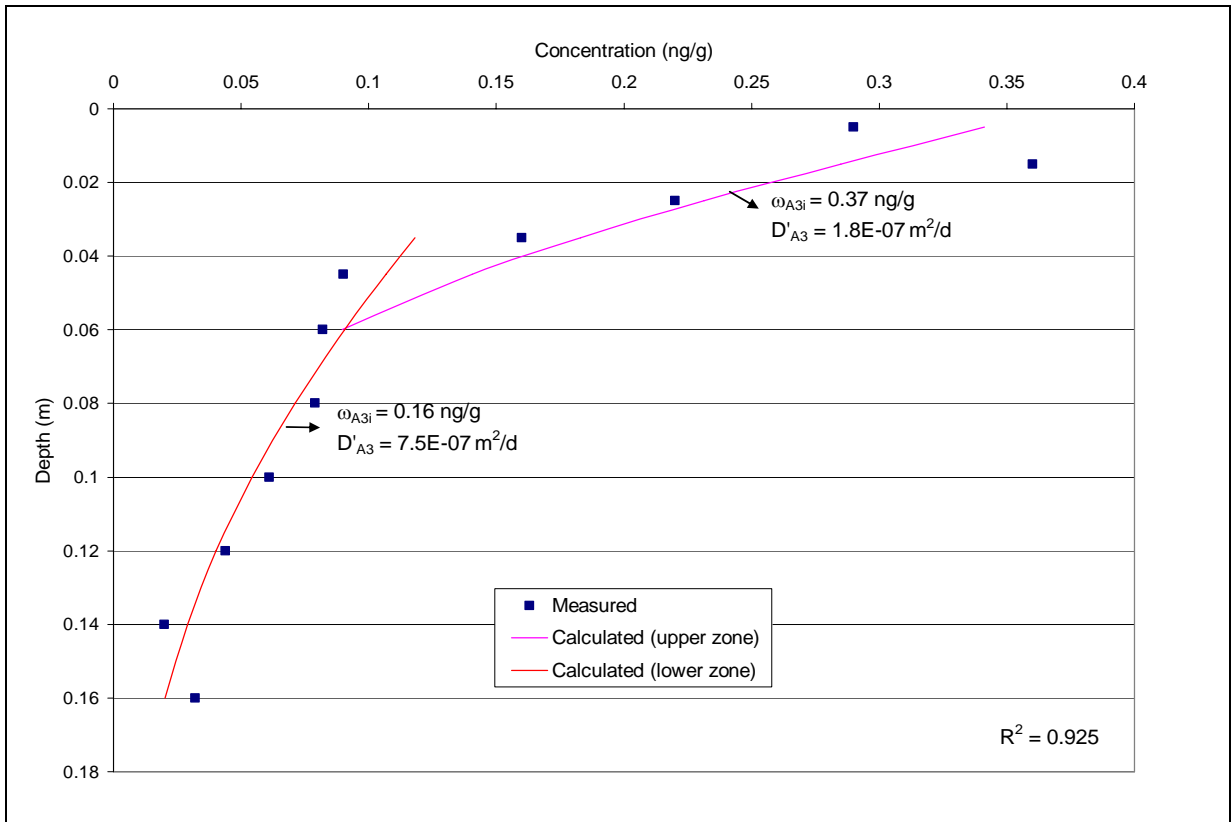


Figure 17. Vertical predicted distribution of PCB-153 in soil

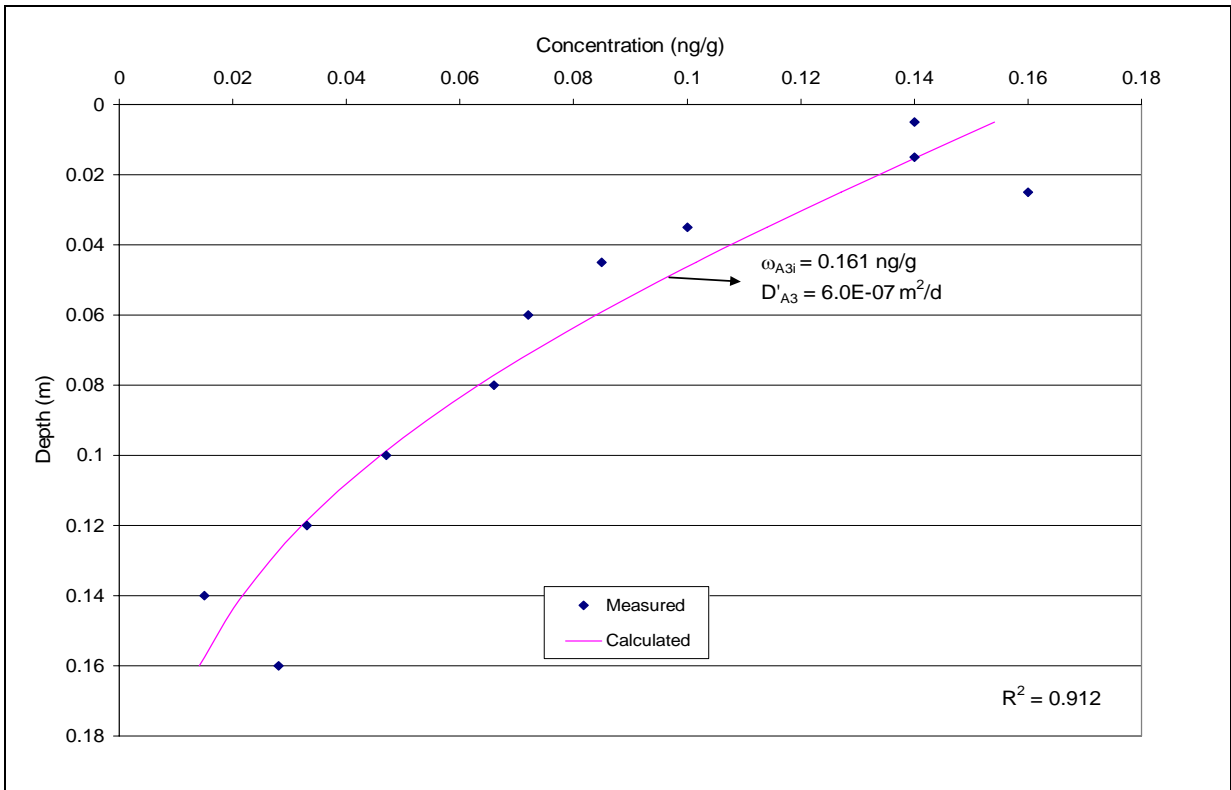


Figure 18. Vertical predicted distribution of PCB-101 in soil

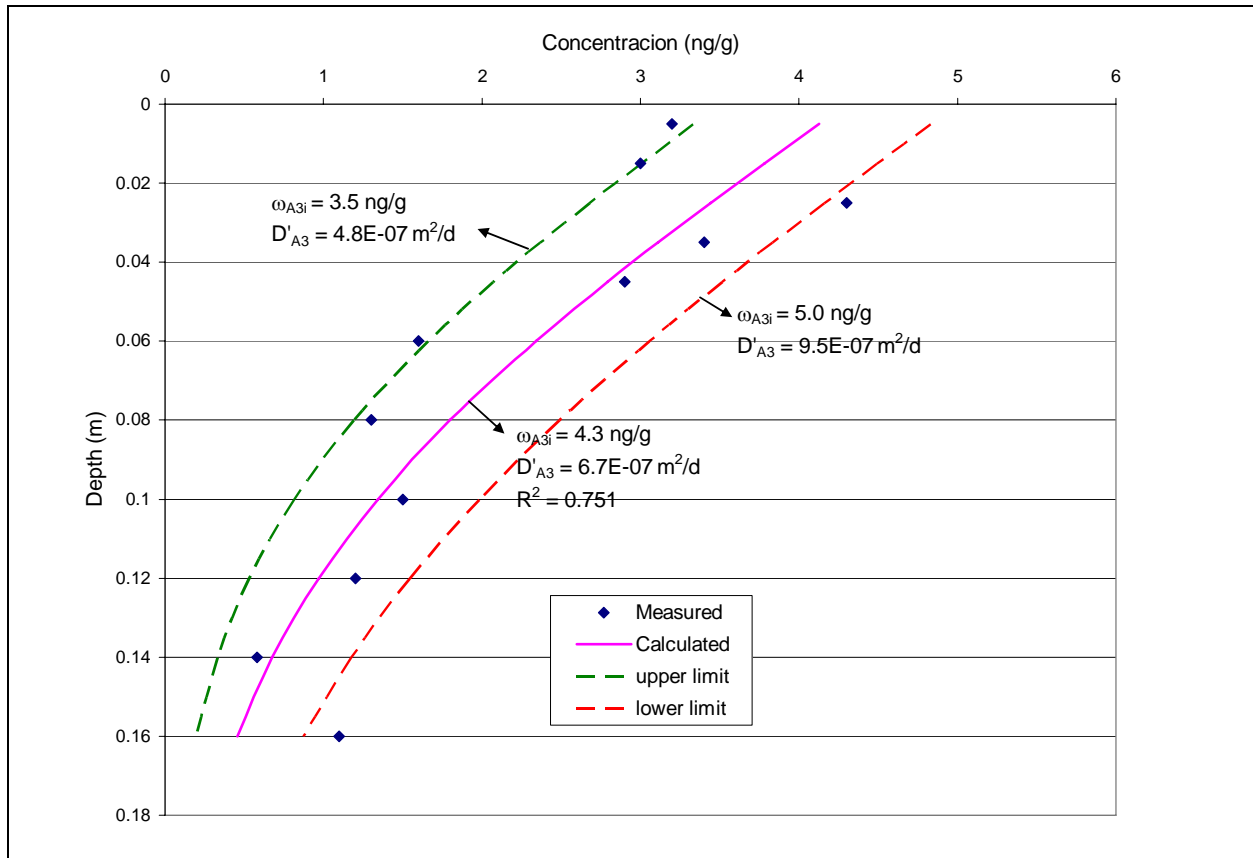


Figure 19. Vertical predicted distribution of 53 PCB congeners

As can be seen on Figure 19, upper and lower limits for the concentration were calculated due to the highly dispersed data for total PCBs.

4.3 Relative Magnitudes of Media Diffusion Coefficients

The above analysis allows estimates of D_{A3} as defined by Equation 21. In this section computations will be performed to estimate all three individual coefficients in this equation. The air and water phase individual values can be calculated using Equations 16 and 17 and inserted into Equation 21. With known D_{A3} values the sorbed phase diffusion coefficients, D_{AS} , can be determined by difference. Once these calculations are completed, the relative magnitudes and individual contributions of each phase diffusion coefficient to the overall process can be obtained.

Diffusion coefficients for air and water are shown in Table 18 along with soil-water partition coefficients for each PCB calculated using Equation 11.

Table 18. Calculated diffusivities and partition coefficients

	PCB-52	PCB-153	PCB-101
D_{A1} (m ² /d)	8.05E-03	8.05E-03	8.05E-03
D_{A2} (m ² /d)	3.11E-06	3.11E-06	3.11E-06
K_{A32} (L/kg)	1535	4196	2538
K_{A32} (m ³ /kg)	1.53	4.20	2.54

The latter partition coefficient, Henry's constant, and soil properties from Table 15, are inserted into Equation 20 to calculate constant K.

Values of D'_{A3} from the model, shown in Figures 16, 17, and 18, and calculated Ks are used to get the effective diffusion coefficients, D_{A3} , for each PCB.

A single value for the effective diffusivity of PCB-153 was found averaging both upper and lower zones, based on the depth modeled by each D'_{A3} parameter. The top 4 cm of the profile, corresponding to 25% of the total depth modeled, was predicted with the lower diffusivity, while the other 75% corresponds to the highest value. Thus, the average was done as follows

$$D'_{A3} = 0.25 \cdot (D'_{A3})_{uz} + 0.75 \cdot (D'_{A3})_{lz} \quad (25)$$

where subscripts *uz* and *lz* stand for upper zone and lower zone, respectively.

Table 19 contains all individual coefficients in Equation 21, including the resultant D_{AS} .

Table 19. Effective diffusion coefficient and its components

	PCB-52	PCB-153	PCB-101
D'_{A3} (m ² /d)	4.40E-07	6.08E-07	6.00E-07
K	1.35E+03	1.35E+03	1.35E+03
D_{A3} (m ² /d)	4.40E-07	6.08E-07	6.00E-07
$D_{A1}H\rho / \rho_b K_{A32}$ (m ² /d)	3.54E-08	9.39E-09	1.82E-08
$D_{A2}/\rho_b K_{A32}$ (m ² /d)	1.50E-09	5.49E-10	9.07E-10
D_{AS} (m ² /d)	4.03E-07	5.98E-07	5.81E-07

Figure 20 illustrates how PCB-52 vertical profile would look like depending on the phase transport process included in the model. The continuous line represents its profile assuming transport of the chemical only in the air phase, in other words neglecting the $D_{A2}/\rho_b K_{A32}$ and D_{AS} terms in Equation 21. Analogously, the dot line corresponds to PCB-52 vertical distribution with transport in the air and water phases. The dash line is the profile considering all three phases in the soil, being the one that fits the measured concentrations.

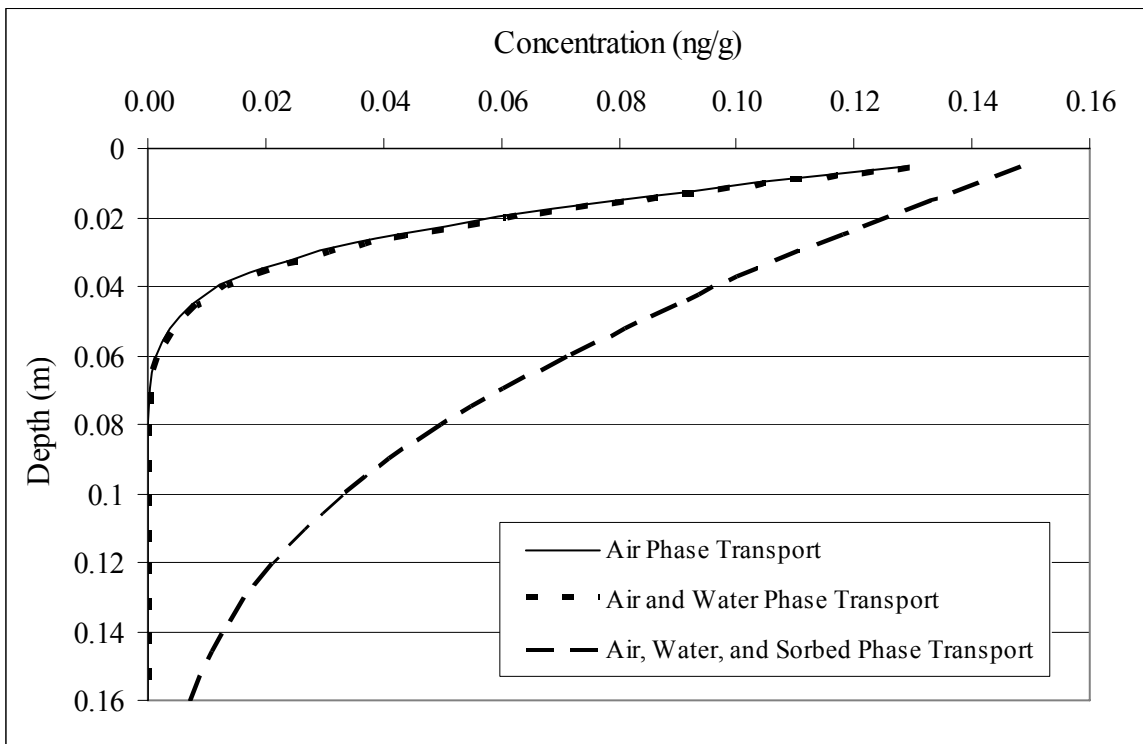


Figure 20. Relative phase transport processes for PCB-52

Clearly, according to Figure 20, the sorbed phase transport process delivers more chemical to the soil. The concentrations are higher and the depth of penetration into the soil column is greater.

Assessment of the relative importance of each phase to the overall diffusion process can be done comparing all three air-, water-, and solid-phase diffusion terms. Percentage contributions of each term are presented in Table 20.

Table 20. Percentage contribution to effective diffusion coefficient

	Contribution to D_{A3} (%)		
	PCB-52	PCB-153	PCB-101
Air	8.05	1.55	3.04
Water	0.34	0.09	0.15
Solid	91.61	98.36	96.81

5. DISCUSSION

5.1 Vertical Distribution of Selected Congeners

The highest measured concentrations for all congeners are located within the top 3 cm of soil (see Table 13), which agrees with the observation that the major input of chemicals to soil is through deposition from air. Therefore higher concentrations should be close to the interface. One of the parameters assumed for the model was the concentration at the air-soil interface, ω_{A3i} , being equal or higher than the maximum concentration from the measured profile. Thus, the model depicts well the distribution suggested by measured profile with concentrations decreasing with increasing depth.

The three numerical values of D_{A3} have the same order of magnitude for all PCBs. However, a trend is observable; there is an increase of this diffusivity with increasing molecular weight. However, since only three congeners are being considered, this cannot be taken as a fact for all PCBs.

The use of polychlorinated biphenyls began several decades ago but a precise time for modeling their distribution is unknown. However, from modeling results the effect of considering a different time can be inferred. It was observed with the given example for PCB-52 that for a time twice as long, D_{A3} was reduced by half. Analogously, if time were multiplied by 4, the resultant D_{A3} would be its value divided by 4. Therefore, considering longer time indicates slower movement of the chemical but gives an effective diffusivity with the same order of magnitude.

Magnitudes of the total square error and R^2 point out that a good model was developed for the selected PCBs. However, a poor fit is observed with the total PCBs concentration including 53 congeners. Properties and concentrations vary among chemicals; therefore it is not

possible to find a single pair of model parameters (ω_{A3i} , D'_{A3}) that provides an ideal fit for the summed concentrations of 53 different isomers. It is sufficient to have obtained an intermediate effective diffusion coefficient with respect to individual PCBs.

Looking at the vertical distribution of PCB-153 in Figure 17, the concentration at the interface for this chemical is much higher than for PCB-52 and PCB-101. A single pair of model parameters would not give a good fit for this profile, therefore two fitting-curves were used, one for the top 4 cm and another for the concentrations going deeper below 4 cm.

On the other hand, the vertical distribution of chemicals in soils not only depends on characteristics of the soil system but also on physical-chemical properties of the contaminant. Therefore, dissimilar behavior between congeners could be in part attributed to their chemical properties. Figure 14 shows similarity in the distribution of all three congeners. The only difference is that heavier PCBs move slightly deeper in the soil than the lighter PCBs. In other words, at any specific depth, below 4 cm, lower concentrations will be detected for lighter congeners, while higher concentrations will correspond to the heavier ones. This agrees with the tendency of having higher D_{A3} values for heavier congeners.

Figure 19 reiterates the fact that different vertical distribution should be expected for different chemicals and, as already mentioned, a single perfect fit can not be found when adding concentrations of various chemicals. This plot can be used as a guide to predict model parameters, in this case more suitable for PCBs.

5.2 Relative Partition between Soil Phases

Heavier PCBs, which are poorly-soluble and non-volatile congeners, have the tendency to remain bounded to organic matter in soils. This can be noticed by the magnitude of soil-water partition coefficients K_{A32} (Table 18). The highest value corresponds to the heaviest congener,

PCB-153, confirming that its concentration in the solid-phase is higher than in water. For that reason, the distribution of this chemical is more likely to be subject to the movement of solid particles in soils.

The soil-water partition coefficients were estimated with the organic carbon partition coefficients, K_{OC} , and the natural organic carbon content of the soil (Equation 11). Therefore magnitudes of K_{A32} show the same relative values as K_{OC} (Table 14). Alternatively, if the soil has higher organic carbon content, a higher fraction of the chemical will be bound to soil particles.

Partitioning of the chemical between soil phases, characterized by the magnitude of its partition coefficients, will influence the dominance of a determined phase transport process. For example, the fate of a chemical with a high Henry's constant will be less influenced by the movement of water in the soil or by the mechanisms through which the sorbed phase or soil particles are transported.

5.3 Dominant Transport Mechanisms

As previously referred, all three effective diffusion coefficients, D_{A3} , are of the same order of magnitude. Table 19 shows key results for this study. Comparison of the magnitudes of all coefficients in Equation 21 allows assessment of the contribution of each soil phase transport process to the overall diffusion process of the chemical. The lower coefficients correspond to the water phase, for all three PCBs. The air phase transport has intermediate values, and the higher values are for the solid phase.

The vertical profiles of PCB-52 showed in Figure 20, including transport either in air or in both air and water phases are almost identical, both lines overlap each other. This is due to the very small value of the diffusion coefficient in the water phase, not making any visible change in

the profile. If the sorbed phase is not included a very steep profile is obtained which is not consistent with the measured field data. Additionally, the maximum depth reached by the chemical would be approximately 0.07 m (7 cm) which do not agree with the fact that concentrations were detected at 16 cm deep. Both air and water transport processes are deduced to be slow compared to sorbed phase transport, which is responsible of having higher concentrations in the soil and reaching greater depths.

Percentage contributions in Table 20 clearly confirm the importance of the soil solid phase in the fate of organic chemicals, and the reason why this phase must be considered when modeling the transport of chemicals in soils. More than 90% of the overall transport process for all congeners corresponds to transport of the sorbed phase.

The most important contribution of the model developed has been to elucidate this relative importance of each soil phase transport mechanism. Since the dominant diffusion coefficient corresponds to the solid phase, the prevailing transport mechanisms are those that enhance the movement of solid particles in the soil, such as bioturbation, mechanical-turbation, cryoturbation, and dryoturbation. Mainly if chemicals are strongly attached or have the tendency to be bounded to soil particles, chemical transport will be subject to those soil mixing processes.

5.4 Biodiffusion vs. Sorbed Phase Diffusion Coefficients

Soils transport/mixing processes, such as bioturbation, mechanical-turbation, cryoturbation, and dryoturbation, are the ones that ultimately control the vertical distribution of chemicals in soils.

In Chapter 2 bioturbation diffusion coefficients were estimated to mathematically describe disturbance by animals and humans. Those biodiffusion coefficients represent the displacement of soil particles over time, which is equivalent to the displacement of the chemicals

sorbed to those soil particles. Therefore, chemical concentration-based estimated diffusion coefficients for the soil sorbed phase (D_{AS}) and the displaced particle-based biodiffusion coefficients (D_3) should be comparable.

Important facts must be remembered before comparing these coefficients. The magnitude of the bioturbation diffusion coefficients D_3 vary depending on several factors, such as species, climate conditions, soil characteristics, and location. As thoroughly explained in Chapter 2, soil turnover rates are highly variable as well as depths reached by different species, both determining biodiffusion coefficients. Additionally, different sorbed phase coefficients (D_{AS}) can be obtained depending on the chemical properties and the soil system. On the other hand, the PCBs concentration profiles modeled were measured in a grassland site, although Cousins and coworkers (1999) do not directly refer to the existence of earthworms at this site, it is expected that biological activity exists in grasslands, dominated by earthworms.

Based on the data presented in Tables 9 and 11, and Figures 7 and 12, bioturbation by earthworms can be characterized by biodiffusion coefficients between $1.6E-08$ and $4.4E-06$ m^2/day (equivalent to 0.02 and 6.0 $cm^2/year$), with an average of $9.18E-07$ m^2/day (3.35 $cm^2/year$). Alternatively, the estimated sorbed phase diffusion coefficients from Table 19 are $4.03E-07$, $5.98E-07$, and $5.81E-07$ m^2/day (equal to 1.47 , 2.18 , and 2.12 $cm^2/year$) for PCB-52, PCB-153, and PCB-101, respectively. Therefore, sorbed phase diffusion coefficients fall within the accepted range describing bioturbation, and both processes are comparable.

Similarly, looking at the data in Tables 10 and 11, and Figures 8 and 12 for all other soil biota species studied, biodiffusion coefficients are between $4.8E-09$ m^2/day (0.02 $cm^2/year$), which is the smallest value corresponding to ants, and $4.9E-06$ m^2/day (18 $cm^2/year$) being the maximum coefficient estimated for gophers. Also in this case, sorbed phase diffusion

coefficients for all three chemicals are within the range for these species, and have the same order of magnitude than the average biodiffusion coefficients estimated for the three biota sub-categories shown on Table 11.

An equivalent comparison with the estimated mechanical-turbation diffusion coefficients would not be as significant as with bioturbation because this is a periodic process, as explained in Chapter 4.

6. CONCLUSIONS AND RECOMMENDATIONS

6.1 Conclusions

- Bioturbation is the most important soil transport/mixing process responsible for the sorbed phase transport of organic chemicals in soils due to its ubiquitous distribution on earth. Under extreme climate conditions, in the coldest and driest regions, or in soils with high shrink-swell potential, cryoturbation and dryoturbation may play a more significant role.
- Earthworms are the main contributors to the bioturbation process. However, other species must be considered depending on location and characteristics of the site.
- Quantification of bioturbation is difficult to achieve due to the extensive variety of contributor species and the wide range of factors that influence their behavior, soil turnover rates, and depths of penetration.
- According to the normal distribution of the data collected on depths for all species, the average depth of soil bioturbation is 20 cm. For earthworms, this average depth is between 10 and 12 cm.
- Bioturbation by humans through tillage practices involves high soil turnover rates, adding periodic disturbance to the continuous soil mixing by animals.
- PCBs are transported principally in association with the soil solid phase. Vertical sorbed phase transport of organic chemicals has more influence in their distribution in soils than transport through the soil air or water phases. The latter two are insignificant in the overall transport process for the chemicals studied.
- The inclusion of transport in the soil solid phase delivers more chemical to the soil and greater depths of penetration than just considering air and water diffusion.

- The effect and contribution of the sorbed phase transport of chemicals is more notorious for heavier PCB congeners, since they bind strongly to soil particles. Therefore, distribution of the heavier PCB congeners is more likely to be subject to the mixing of soil by bioturbation and other mixing mechanisms.
- Fate and transport models for organic chemicals in soils must take into account vertical transport in the sorbed phase, quantitatively linked with bioturbation through characteristic biodiffusion coefficients.
- There is an agreement between values found for the chemical concentration-based estimated sorbed phase diffusion coefficients (D_{AS}) and the displaced particle-based bioturbation diffusion coefficients (D_3).

6.2 Recommendations

- Experimental measurements of soil turnover rates should be done for different earthworm species and other animals in varied types of soil, since there is limited data published in this field.
- The model developed should be fitted to measured concentration soil profiles for other persistent chemicals, such as heavy metals and radionuclides, to obtain more data on D_{AS} values. Also, collection of soil profiles in agricultural soils is recommended.
- For modeling of profiles in specific sites and to improve accuracy, it is recommended to consider the variation of soil properties with depth in the model, specifically for soil organic content and moisture content.

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