The Impacts of Grazing Animals on the Quality of Soils, Vegetation, and Surface Waters in Intensively Managed Grasslands

G. S. Bilotta,[1,2] R. E. Brazier[1] and P. M. Haygarth[2]

[1]Department of Geography, University of Exeter, Exeter, Devon EX4 4RJ, United Kingdom
[2]Cross Institute Programme for Sustainable Soil Function (SoilCIP), Institute of Grassland and Environmental Research (IGER), North Wyke Research Station, Devon EX20 2SB, United Kingdom

Advances in Agronomy, Volume 94

0065-2113/07 $35.00
DOI: 10.1016/S0065-2113(06)94006-1

This chapter provides a comprehensive review of the literature relating to the impacts of grazing animals on the quality of soils, vegetation, and surface waters. It focuses on intensively managed grasslands where there is the greatest potential for these impacts to be observed. The chapter indicates that while well-managed grazing can be beneficial to the environment, intensively managed grazing can actually lead to the degradation of both the soil and vegetation of grassland environments. The various causes, forms, and consequences of this degradation are discussed in detail, and gaps in the knowledge are identified. The chapter highlights the need for recognition and quantification of the relationships between the on-site impacts of grazing animals (i.e., changes in soil properties and vegetation cover) and the off-site impacts of grazing animals (i.e., the impact of these changes on hydrology and water quality in surface waters), as these relationships have, in the past, only been alluded to by authors. However, there exists relatively little research evidence to support and quantify these relationships, thus herein we describe data required to address the lack of understanding of the role of grazing animals on grasslands. Finally, the last section of this chapter considers the land management and remediation options available for the reduction of the impacts of intensive livestock farming.

I. INTRODUCTION

Grasslands cover a large portion of the temperate landmass, including significant areas of Europe, North America, New Zealand, and Australia. In Western Europe, for example, grasslands occupy almost 40% of the agricultural area, although this proportion is even higher in some of the countries within Europe (e.g., Austria, 57%; Ireland, 76%; Switzerland, 72%; United Kingdom, 65%) (Peeters, 2004). Much of this grassland is grazed by livestock, providing its people with meat and dairy products, employment, and a source of income which in some regions is central to the local economy (Carroll *et al*., 2004; Reynolds and Frame, 2005). While well-managed grazing can be beneficial to the environment, enhancing nutrient cycling and promoting floral and faunal biodiversity (Isselstein *et al*., 2005; Pykala, 2000; Rook and Tallowin, 2003), it is now recognized that intensively managed grazing can lead to environmental degradation (Evans, 1997, 1998; Heathwaite *et al*., 1990; Kellett, 1978; Mulholland and Fullen, 1991; Patto *et al*., 1978; Trimble and Mendel, 1995). Intensively managed grasslands tend to be located in lowland areas and are characterized by high stocking densities (i.e., high number of animals per unit area) and high inputs of chemical (e.g., fertilizers, pesticides, imported animal feeds) and energy (e.g., farm machinery, tractors) resources. These practices are designed to maximize agricultural productivity and are fairly widespread in temperate regions, with ~29% of the total land area in England in Wales managed in this manner (Defra, 2005). However, intensive farming practices have been associated with changes in the percentage cover and biodiversity of grassland vegetation and alterations of the condition of the grassland soil. Together, these changes have been linked to a modified hydrological behavior of pastures and, ultimately, the deterioration of water quality in surface waters within these environments (Kurz *et al*., 2006; McDowell *et al*., 2003; Monaghan *et al*., 2005).

The environmental degradation induced by grazing animals is a consequence of several key activities which livestock carry out, including defoliation, treading, and excretion. First, excessive defoliation by grazing animals and damage to plant tissues as a result of direct (e.g., crushing, bruising, shearing) and indirect (e.g., changes to the rhizosphere as a result of compaction, pugging, and poaching) treading effects can reduce both the biodiversity of the pasture and the percentage cover of the vegetation (Matches, 1992). This can lead to a decline in faunal biodiversity and pasture productivity, and may eventually produce bare patches within the pasture where the soil surface is exposed to erosive agents (Edmond, 1958; Evans, 1998; Matches, 1992). Second, the treading action of livestock hooves on the soil surface, particularly on wet and saturated soils, can cause structural deformation of the soil such as compaction, pugging, and poaching, which can reduce the porosity of the soil and increase the bulk density of the soil (Climo and Richardson, 1984; Di *et al*., 2001; Drewry and Paton, 2005). This in turn can decrease the infiltration capacity (Mulholland and Fullen, 1991) and hydraulic conductivity of the soil (Greenwood *et al*., 1997; Willatt and Pullar, 1983), and can therefore promote surface runoff generation (Di *et al*., 2001; Heathwaite *et al*., 1990). If this occurs over large areas of a catchment, it can alter the hydrology of whole rivers (Carroll *et al*., 2004; Harrod and Theurer, 2002). Indeed, there is anecdotal evidence to suggest that the response of rivers to rainfall events is becoming more intense because of poor agricultural soil management practices (Reynolds *et al*., 2002), and a report by the UK Environment Agency (2002) estimated that the damage to homes, commercial property, and agricultural land resulting from the poor soil structure caused by intensive agriculture costs the UK approximately £115 million per annum (in 2000 prices). Furthermore, this enhanced runoff can potentially mobilize large amounts of sediment and colloidal material (including soil, plant, and livestock fecal matter) from the damaged and exposed soil surface of the grassland, and deliver this matter into surface waters where it could contribute to sedimentation problems (Harrod and Theurer, 2002; Walling *et al.*, 2003), eutrophication (Haygarth and Jarvis, 1999; Heathwaite and Johnes, 1996), and pathogenic contamination (Chadwick and Chen, 2002; Oliver *et al.*, 2005b). The deterioration of water quality induced by intensive grazing is of particular concern to those involved in meeting the demands of environmental legislation such as the EU Water Framework Directive (Neal and Jarvie, 2005).

The environmental degradation of grasslands induced by grazing animals has become more prominent over the last few decades and has been associated with the intensification of agricultural production and the gradual decline in the area under grassland, while livestock numbers have been maintained or have increased, resulting in higher stocking densities on the remaining grassland area (Evans, 1997; Heathwaite *et al.*, 1990). This chapter provides a review of the literature on the effects of grazing animals on the quality of soils, vegetation, and surface waters in intensively managed, temperate grasslands, synthesizing the key findings, and identifying those areas where further research is required.

II. IMPACT OF TREADING BY GRAZING ANIMALS ON GRASSLAND SOILS

Soil is composed of inorganic and organic primary particles, the size distribution of which determines the texture of the soil. Primary particles may be bound together to form aggregates, the size and arrangement of which determine the volume and configuration of spaces and pores within the soil and constitute collectively, the structure of the soil. The resistance of this soil structure to an imposed force (i.e., the shear strength of the soil) results from internal friction and interlocking of primary particles and aggregates, supplemented by inter-particulate cohesion and bonding (Patto *et al.*, 1978). When a load imposed on the soil (i.e., a shear stress) is greater than the load-bearing capacity of the soil, it will lead to a modification of the structural configuration (i.e., soil deformation). Grazing animals can exert a large amount of force on the soil surface due to their large weight and relatively small hoof area. The amount of pressure exerted on the soil is dependent on the species and age of the grazing animal. The amount and form of soil structural alteration (i.e., deformation) which occurs as a result of this force is primarily determined by the stocking density, soil moisture content, soil texture, and the presence/absence of a protective vegetation cover. This section of the chapter first examines the key factors that influence the amount and form of soil deformation, and then moves on to discuss the individual forms of soil deformation. This is then followed by an evaluation of the likely implications that these changes may have on the wider environment.

III. FACTORS INFLUENCING THE AMOUNT AND FORM OF SOIL STRUCTURAL ALTERATION

A. Animal Species and Age

The force imposed on the soil by a grazing animal is a function of the weight of the animal and the area of contact between the animal hooves and the soil surface (Patto *et al.*, 1978). Clearly, this will vary depending on the species and age of the animal, with cattle exerting the greatest forces onto the soil. For example, an adult cow weighs ~350–600 kg (Abdel-Magid *et al.*, 1987; Mulholland and Fullen, 1991; Scholefield and Hall, 1986; Wind and Schothorst, 1964). When a cow is static, this mass will be distributed over four hooves, each with an area of around 60–90 cm^2 (Frame, 1971), depending on cow breed and age (Scholefield and Hall, 1986). This creates static pressures of around 200 kPa (Di *et al.*, 2001; Willatt and Pullar, 1983). These forces may be significantly increased when the animal is walking and has only two or three hooves in contact with the ground at any one time (Di *et al.*, 2001; Trimble and Mendel, 1995; Willatt and Pullar, 1983), leading to forces of up to 400 kPa (Climo and Richardson, 1984). The forces may increase again if the hoof is not placed onto a flat surface (Di *et al.*, 2001; Willatt and Pullar, 1983). For sheep, the body mass and resultant hoof forces are much lower than those for cattle, ranging from 50 to 80 kPa while static (Noble and Tongway, 1986; Willatt and Pullar, 1983), and up to 200 kPa when moving (Willatt and Pullar, 1983). Nevertheless, both the forces from sheep and cattle exceed those recorded from tractors which are known to cause compaction, yet exert forces ranging from just 30 to 150 kPa (Soane, 1970; Soehne, 1958). Moreover, the soil compaction caused by grazing animals is likely to be more widespread within paddocks than that caused by vehicle tracks (Drewry, 2006).

The force imposed by an animal hoof can be divided into two components; a normal component acting vertically downward onto the soil and a tangential component acting horizontally to the soil surface (Patto *et al.*, 1978). It is this normal component of force, which occurs as the hoof is placed down onto the soil surface, which is responsible for soil compaction, pugging, and poaching (Greenwood and McKenzie, 2001). The tangential component tends to occur as the hoof is lifted for a subsequent step and causes a shearing of the soil which can smear the soil and tear vegetation (Alexandrou and Earl, 1997).

B. Stocking Density

Several researchers have found that the amount of soil structural alteration induced by grazing animals increases with stocking density (the number of animals per unit area) (Bryant *et al*., 1972; Langlands and Bennett, 1973; Mulholland and Fullen, 1991; Willatt and Pullar, 1983). This has been attributed to the fact that as the stocking density increases, the frequency at which any given point in the paddock/pasture is visited by a grazing animal also increases. Each time a point is revisited, it leads to further breakdown of soil structure and water-stable aggregates, making the soil more susceptible to further alteration (Patto *et al*., 1978; Wind and Schothorst, 1964). Kellett (1978) proposed that increased soil structural alteration with increasing stocking density is also a result of the lower vegetation/protective cover available at higher stocking densities, where there are greater rates of defoliation. In contrast to the above findings, a 30-year study by Greenwood *et al*. (1997) found no evidence of increased structural alteration with increased stocking density. Greenwood *et al*. (1997) propose that this may be because the effects of grazing animals are cumulative and therefore tend to reach a common state over the long-term.

The magnitude of the relationship between soil damage and stocking density reported in the literature varies significantly between studies. This may be due to differences in the soil health indicator measured (e.g., bulk density, infiltration capacity, porosity, or macroporosity), differences in the methods by which grazing intensities were simulated/produced, differences in soil type, topography, and climate, differences in stocking management, and differences in the duration of experiments and observations. Trimble and Mendel's (1995) review of the literature revealed that there are strong differences in the methodologies used in livestock impact studies. For example, some studies are carried out on natural watersheds using natural rainfall events, while others utilize small plots and/or flumes and artificial rainfall (Trimble and Mendel, 1995). In one case, a storm equivalent to the 150-year return period was required to produce overland flow from poached land—reducing the confidence in the results (Trimble and Mendel, 1995). Perhaps more importantly, few simulation studies calibrated their means of simulating the effect of livestock with the real animals (Trimble and Mendel, 1995), with the notable exceptions of workers such as Scholefield and Hall (1986). Furthermore, when researchers used outside plots/catchments, there was little consideration of the influence that prior land treatment, such as history of grazing, could have on the results (Trimble and Mendel, 1995). There is a paucity of knowledge about such lag effects, but they may be significant and may persist for decades (Greenwood *et al*., 1997). Trimble and Lund (1982) and Trimble (1988, 1990) suggest that the effects of land abuse and land recovery may take anything from several years to several decades to manifest themselves.

In the existing literature describing the effects of grazing intensity/stocking density on grasslands, it is clear that there are no universal definitions of treatments in terms of stocking rates, duration, and seasonality (Trimble and Mendel, 1995). This can create difficulties when attempting to make cross-study comparisons (Trimble and Mendel, 1995). One standardized measure of stocking density is Livestock Units (LSU) per hectare (Carroll *et al*., 2004). For example, using this definition, a dairy cow is equivalent to 1 LSU, whereas a sheep is regarded as 0.15 LSU (Carroll *et al*., 2004). However, Evans (1998) argues that when stocking density is given in terms of LSU, the comparison is made even more difficult, because it is not known how many animals are grazing an area, for the proportions of the different kinds of animals may vary but give the same LSU intensity. Evans (1998) suggests that if grazing intensity is given in terms of the animal, generally cattle or sheep, intensities can be compared between localities. There is also an issue over the style of livestock management, that is continuous stocking versus rotational stocking. Some authors include the length of grazing time per stocking density, others just mention the stocking density without a reference to the grazing period. This can lead to discrepancies in findings from each study.

The majority of the existing research into the impact of livestock and stocking densities on grasslands has taken place in New Zealand, Australia, and America (Carroll *et al*., 2004). At present, there are very little quantitative data available for other countries such as the United Kingdom (Carroll *et al*., 2004; Trimble and Mendel, 1995). While the data from outside the country of interest can be useful, it must be used with caution as there may be issues of transferability due to differences in climate, soil type, vegetation, and grazing management style (Trimble and Mendel, 1995). Since it is estimated that grasslands cover around two-thirds of the UK land area (Carroll *et al*., 2004; Reynolds and Frame, 2005; Waters, 1994), the impact of stocking density is an important area that requires further research.

C. Soil Moisture

Several authors have reported that soil damage induced by grazing animals is worsened by increasing the moisture content of the soil (Climo and Richardson, 1984; Mulholland and Fullen, 1991; Scholefield and Hall, 1986; Wind and Schothorst, 1964). As mentioned previously, the resistance of a soil structure to an imposed force results from internal friction and interlocking of primary particles and aggregates, supplemented by interparticular cohesion and bonding (Patto *et al.*, 1978). Under soil conditions of low suction or positive water pressure, the effective contact between soil particles (a source of internal friction) is reduced and particles can move more easily along failure planes, reducing soil strength and resistance to structural alteration (Patto *et al.*, 1978). The soil moisture content will also determine the dominant form of soil structural alteration. As a general rule, soil compaction tends to dominate at low to medium soil moisture contents, followed by pugging at the medium to high moisture contents, and poaching on saturated soils (Mulholland and Fullen, 1991).

Scholefield and Hall (1985) suggest that although there is a general assumption that the extent of soil deformation is primarily determined simply by the soil water content, their research supports a more complex model. Scholefield and Hall (1985) found that over a wide range of water contents, deformation due to treading was independent of water content. These results and those of other studies (Mullen *et al.*, 1974) support a model for poaching alluded by Mullins and Fraser (1980) in which soil strength declines progressively during repeated treading in the presence of free water (i.e., water on and within the soil surface, not held in soil pores). Therefore, with this model, the amount of deformation will be determined by the rate of loss of soil strength during treading in wet weather. This rate will be determined by the rate at which water can be incorporated into the soil (which in turn is determined by soil texture and the amount of existing deformation), and also by the sensitivity of interparticular bonds (which in turn is determined by soil mineralogy and organic content) to mechanical disturbance, neither of which may be predicted by a single measurement of the initial state of the soil. This highlights the importance of soil texture and mineralogy, as well as the number of treading instances during wet weather, not simply initial soil moisture content.

D. Vegetation Cover

The protective role of plant cover with respect to damage by animal hooves on the soil has long been recognized (Climo and Richardson, 1984; Kellett, 1978; Scholefield and Hall, 1986). The protection to the soil offered by plants is derived in several ways; first, the above-ground plant matter provides a direct physical boundary between the hooves and the soil (O'Connor, 1956). Second, the below-ground plant matter (roots and stolons) in the soil acts to increase the shear strength of the soil and its load-bearing capacity (Patto *et al.*, 1978). Third, plants provide protection for the soil indirectly through the decomposition of plant residues which bind with the mineral component of the soil and together with other agents, such as calcium/magnesium carbonates, iron/aluminum oxides, and silicates, give rise to water stable aggregates which are more resistant to deformation (Patto *et al.*, 1978; Taylor and Ashcroft, 1972; Tisdale and Oades, 1982).

The degree of protection offered by vegetation depends on the quality and quantity of vegetation (Climo and Richardson, 1984). Established dense turf mats, which often form under low fertility conditions, provide good physical protection to the soil (Sears, 1956). Higher producing pastures, where fertilizer use is more common and the sward is dominated by species such as perennial ryegrass (*Lolium perenne*) and white clover (*Trifolium repens*), tend to be more open and allow direct hoof/soil contact and so offer a lower degree of protection than a dense nutrient-poor grassland sward (Climo and Richardson, 1984). It is well known that grassland species vary in their ability to survive trampling and, therefore, in the degree of protection that they provide for the soil surface (Patto *et al.*, 1978). For example, a study in New Zealand by Edmond (1962) found perennial ryegrass to be far more tolerant of treading than white clover.

IV. FORMS OF SOIL STRUCTURAL ALTERATION RESULTING FROM TREADING BY GRAZING ANIMALS

There are three main forms of structural alteration associated with grazing animals, these are: compaction, pugging, and poaching. These forms have sometimes been bulked together by authors and referred to as "poaching" (Kellett, 1978). However, in this chapter, the impacts of grazing animals are treated as individual processes because they operate under different conditions and can have different effects.

A. Soil Compaction

Soil compaction has traditionally been defined as the compression of an unsaturated soil body resulting in a reduction of the fractional air volume (Hillel, 1980). The potential for grazing animals to cause soil compaction was first noted by authors such as Tanner and Mamaril (1959) and Federer *et al.* (1961). Soil compaction occurs when the load of a grazing animal imposed on an unsaturated soil is greater than the load-bearing capacity of the soil. Compressive deformation or soil compaction is illustrated in Figs. 1 and 2. During compaction, particles are forced closer together by the applied load reducing the total pore space and permanently expelling air or water from the soil pores (Patto *et al.*, 1978). This has a number of implications for soil hydrology and vegetation growth, as discussed in Section V.

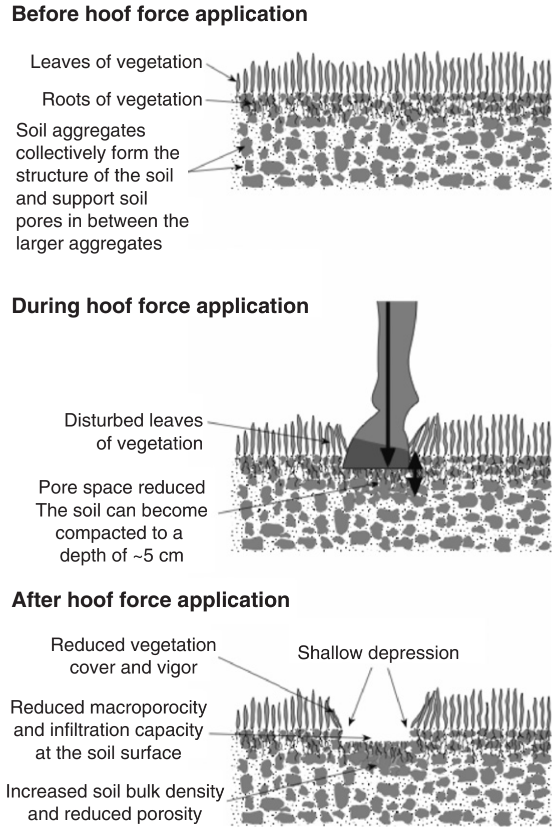

Figure 1 A schematic diagram illustrating the process of soil compaction through the hoof force of grazing animals.

B. Soil Pugging

Soil pugging is the term used to describe the process by which livestock tread on wet soft soil and create deep hoof imprints (Drewry, 2006). This is illustrated by Figs. 3 and 4. Pugging is a type of plastic deformation which occurs on soils with a medium soil moisture content when the animals' load exceeds the bearing capacity of the soil (Patto *et al.*, 1978). During plastic deformation, particles move relative to each other taking up new equilibrium positions with or without a reduction in total pore volume, although the proportion of fine pores will often be increased (Patto *et al.*, 1978). The hoof imprints created by this deformation leave a very uneven pasture surface and can also influence soil hydrology and plant growth, as discussed in Section V.

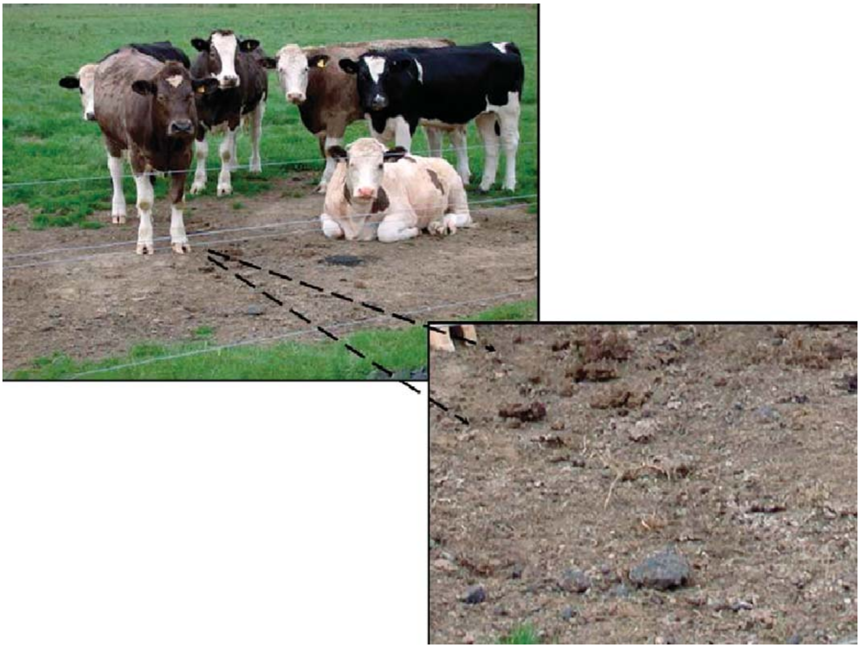

Figure 2 A photograph of a compacted soil under intensively managed grassland in Devon, United Kingdom. Photograph by G. S. Bilotta.

The factors controlling the amount of damage to pasture by pugging are the same as those for compaction (soil moisture content, stocking density, soil texture, vegetation cover) and the mechanisms by which these factors determine the amount of damage have already been discussed in the previous sections and so will not be reexamined. Pugging occurs at soil moisture contents intermediate to those at which compaction (low moisture) and poaching (high moisture) occur. However, soils with high clay contents can behave in a plastic manner even at lower soil moisture contents, making them particularly susceptible to pugging damage (Kellett, 1978).

C. Soil Poaching

Poaching is the term used to describe the slurry-like soil conditions that occur on very wet soil when trampled by livestock (Drewry, 2006). This is illustrated in Figs. 5 and 6. Poaching is a type of elastic deformation which occurs when the animals load exceeds the load-bearing capacity of the saturated soil and the hooves penetrate the soil surface (Kellett, 1978;

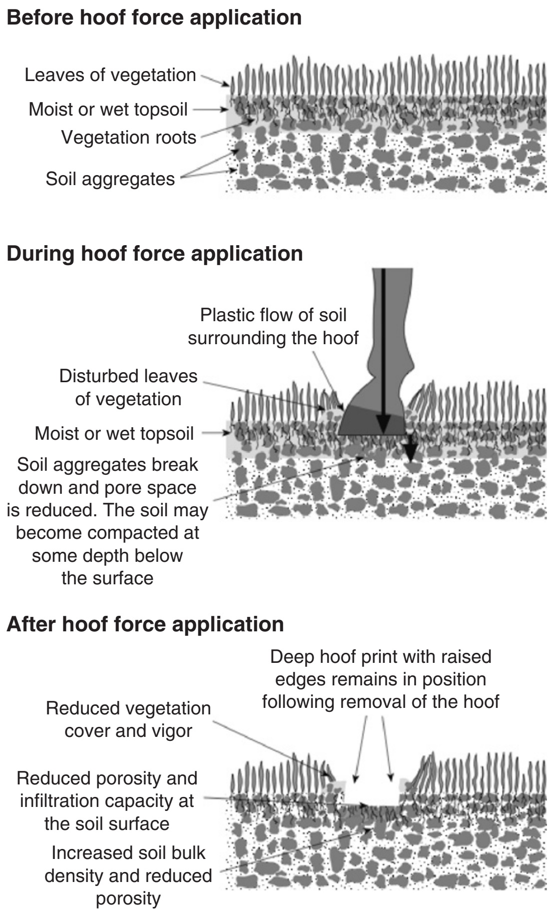

Figure 3 A schematic diagram illustrating the process of soil pugging by grazing animals.

Patto *et al*., 1978). Elastic deformation is associated with lateral bulging of the soil and usually occurs in soils with a high proportion of water-filled pores (Patto *et al*., 1978). Since water is relatively incompressible, the soil recovers without an appreciable change in volume (Patto *et al*., 1978). In some cases, water held in the soil pores is forced from the soil as pressure is applied, but on removal of the load, the water is drawn back between the particles and elastic recovery occurs (Harris, 1971).

Kellett (1978) suggests that the two major factors controlling the amount of damage to pastures as a result of poaching are: (1) the moisture content and (2) the stocking density. The mechanisms by which these factors determine the amount of damage have already been discussed in the previous

Figure 4 A photograph of a pugged soil under intensively managed grassland in Devon, United Kingdom. Photograph by G. S. Bilotta.

section and so will not be reexamined. Poaching can be extremely disruptive to plant growth and also has serious implications for soil hydrology, as discussed in Section V.

V. THE IMPACTS OF SOIL STRUCTURAL ALTERATION BY GRAZING ANIMALS

A. Treading and Soil Physical Properties

Changes to soil physical properties caused by grazing animals have received little attention compared with compaction of arable soils, despite the serious implications and the fact that in contrast to arable soils, there is little opportunity to ameliorate compacted soil through tillage in permanent pasture (Greenwood and McKenzie, 2001). A number of authors have reported that the effect of stock treading and the resultant deformation tends to be confined to the upper layers of soil, within ~50 mm of the soil

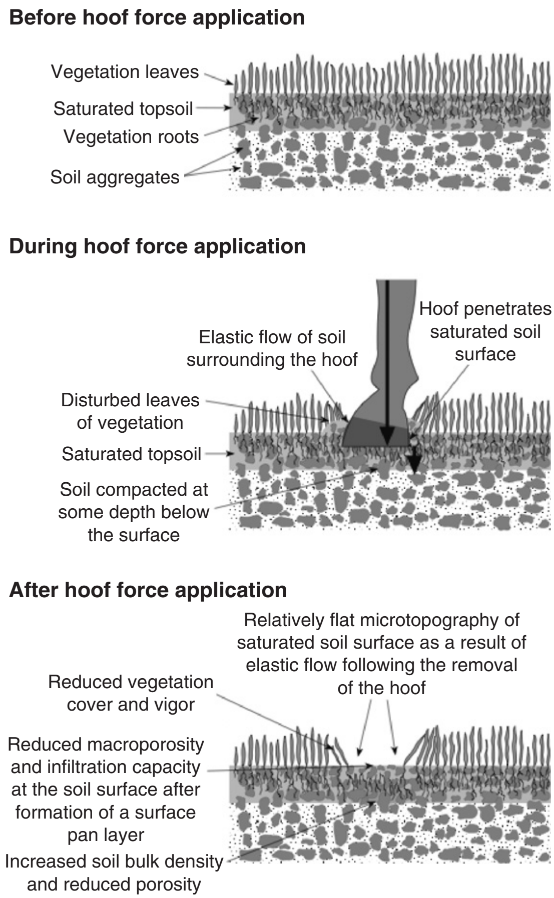

Figure 5 A schematic diagram illustrating the process of soil poaching by grazing animals.

surface (Alderfer and Robinson, 1947; Climo and Richardson, 1984; Drewry, 2006; Greenwood *et al.*, 1997; Mulholland and Fullen, 1991; Tanner and Mamaril, 1959). Alderfer and Robinson (1947) argue that it is this zone which is very important in determining soil hydrology and plant growth/vigor. One of the main effects of soil compaction by grazing animals is the reduction in pore space and macroporosity which has often been associated with an increase in bulk density (Climo and Richardson, 1984; Di *et al.*, 2001; Drewry and Paton, 2005; Mulholland and Fullen, 1991; Willatt and Pullar, 1983). The reduction in pore space is known to influence soil hydrology, vegetation growth, and the vitality of soil fauna, these impacts are discussed in the following sections. Soil pugging incorporates the effects of soil compaction, as well as an alteration of soil surface microtopography, creating a

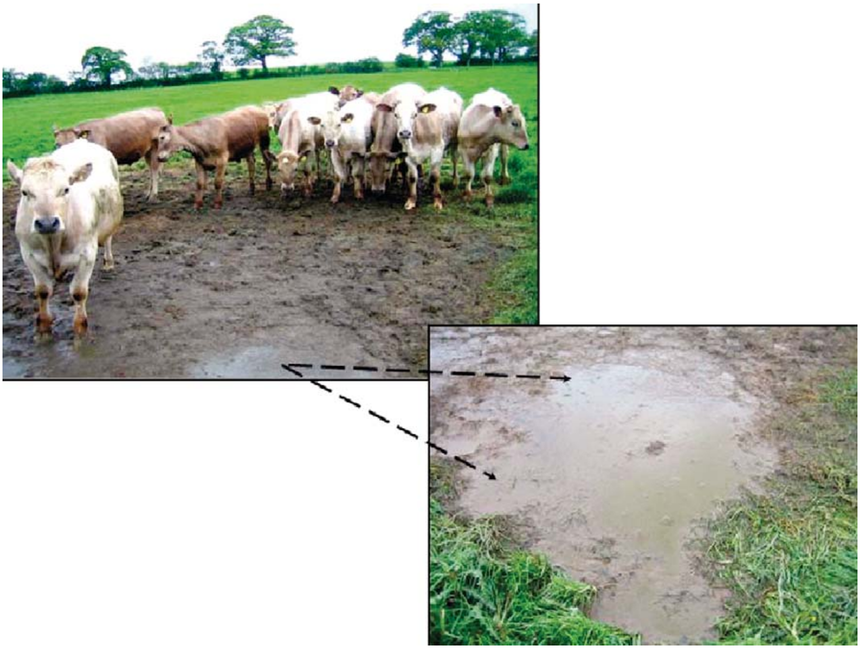

Figure 6 A photograph of a poached soil under intensively managed grassland in Devon, United Kingdom. Photograph by G. S. Bilotta.

rough and uneven surface. Soil poaching is associated with a breakdown of soil aggregates and a rearrangement of soil particles in the surface soil horizons. This can lead to an increased bulk density and may also lead to the formation of a surface pan as the soil dries (Kellett, 1978). O'Connor (1956) suggests that it is also possible, when the soil moisture content is high, for hoof forces to greatly exceed the bearing strength of the soil and thus for hooves to penetrate the soil surface and cause large increases in soil bulk density at a greater depth in the soil, compressing the soil at some depth below the surface.

B. Treading and Soil Hydrology

Soil compaction and the reduction in pore space can lead to decreases in both the hydraulic conductivity of the soil (Greenwood *et al.*, 1997; Willatt and Pullar, 1983) and the infiltration capacity of the soil (Alderfer and Robinson, 1947; Heathwaite *et al.*, 1990; Mulholland and Fullen, 1991). This can make the soil more prone to ponding, thus rendering the soil more susceptible to further deformation such as poaching (Mulholland and Fullen, 1991; Patto *et al*., 1978). Furthermore, it can make the soil more prone to surface runoff generation as drainage becomes impeded. For example, Heathwaite *et al*. (1990) found that infiltration capacity was reduced by 80% and surface runoff volumes were increased by nearly 12 times on heavily grazed grassland compared with ungrazed grassland. Mulholland and Fullen (1991) showed that infiltration rate was very sensitive to the soil structural change caused by stock treading and showed a 98.5% decrease in infiltration in heavily trampled areas, although the largest decrease in infiltration occurred on initial compaction (87.5%), with only minor decreases for subsequent compactions.

Soil compaction is not the only form of deformation that can influence soil hydrology. For example, soils that have undergone pugging may be more prone to ponding in surface depressions. Similarly, soils that have undergone poaching and the formation of a surface-pan may also be more susceptible to ponding and the generation of surface runoff. These changes to the soil hydrology have implications for runoff from grazed land, potentially modifying not only the quantity of runoff, but also the quality of runoff, in terms of sediment and nutrient loads moving over and through the soil (Di *et al*., 2001; Heathwaite *et al*., 1990). The relevance of the changes in soil properties as a result of treading along with the effects of other activities carried out by grazing animals is illustrated in Fig. 7. These changes to the soil physical properties can also influence nutrient transformation processes within the soil by altering the moisture regime and affecting soil redox potential and plant uptake processes (Climo and Richardson, 1984; Di *et al*., 2001).

C. Treading and Vegetation Growth

Several workers have reported that treading by grazing animals can cause a significant reduction in herbage growth/yield (Cluzeau *et al*., 1992; Edmond, 1962; Federer *et al*., 1961; Matches, 1992; Tanner and Mamaril, 1959). It has been estimated that the reduction in herbage yield may be as large as 25–40% (Carter, 1962; Muller, 1965; Schothorst, 1963), made up of 5–20% from immediately damaged and buried herbage, and 10–20% from reduced production by the remaining damaged sward (Kellett, 1978). Tanner and Mamaril (1959) reported a 20% decrease in herbage yields as a result of soil compaction which caused a severe decline in soil pore space and aeration in the root zone. Cluzeau *et al*. (1992) found that direct plant damage during stock trampling was responsible for the destruction of a large amount of plant material and a reduction in herbage yield. Although pasture grasses are particularly well adapted to frequent harvesting of their leaves, possessing condensed growing points that are close to the base of the plant, the action of

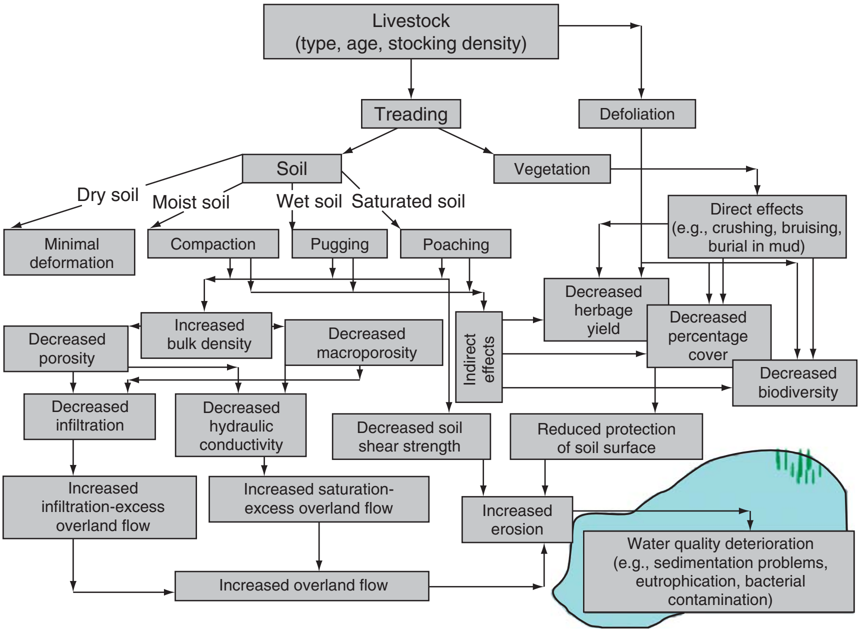

Figure 7 A conceptual diagram representing the environmental degradation induced by overgrazing (treading and defoliation).

poaching can damage these growing points and bury plant structures in mud (Kellett, 1978). In addition, grass roots can be seriously affected by poaching as these tend to be concentrated in the top 50 mm of the soil—the zone of poaching damage (Kellett, 1978). However, it has been difficult to determine the extent to which pasture response occurs due to changes in soil physical condition caused by soil compaction, pugging, and poaching alone, or due to trampling on the plant matter directly, leading to plant bruising, crushing, root damage, and plant displacement or burial in mud. A study by Drewry *et al.* (2001), which simulated dairy cow treading in controlled field conditions allowing for soil compaction, but not pugging and with minimal plant damage, found that significant reductions in pasture yield could be induced by compaction alone (i.e., indirect effects). Nevertheless, in the real field situation, reductions in plant growth are probably a result of a combination of direct and indirect effects of treading and soil compaction (Di *et al.*, 2001). As well as the reduction in grassland vegetation yield due to treading effects, the uptake of plant matter by grazing animals may also be reduced. This is because grazing animals will often refuse herbage when it is rendered unpalatable by contamination with mud or when herbage is broken and left lying on the soil surface (Kellett, 1978).

D. Treading and Soil Fauna

It has been noted that soil fauna generally have positive effects on the soil by: (1) increasing the porosity and permeability, (2) improving soil structure, and (3) enhancing nutrient cycling and soil fertility (Trimble and Mendel, 1995). Earthworms (*Lumbricina*) are often viewed as the most beneficial fauna in terms of improvement of the soil structure (Piearce, 1984; Trimble and Mendel, 1995). However, some workers have indicated that soil deformation by grazing animals can cause a decline in earthworm numbers (Cluzeau *et al*., 1992; Drewry and Paton, 2005; Knight, 1992; Piearce, 1984). Earthworms have difficulty surviving in impacted soil conditions resulting from heavy grazing, although it has not yet been established whether this is due to direct impacts (i.e., mortality due to crushing) or indirect impacts (i.e., sublethal effects due to environmental changes within the soil) (Cluzeau *et al*., 1992; Trimble and Mendel, 1995). Piearce (1984) proposed that there are several mechanisms for the reduction in earthworm numbers in trampled areas. First, trampling can cause death by direct crushing of individuals in the soil surface. Second, a less immediate but possibly equally important mechanism is associated with the reduction in soil porosity which impairs the movement of water and air through the soil and impedes earthworm locomotion. Third, trampling and defoliation can alter the amount of vegetation and hence the quality and quantity of food available for the soil fauna. Finally, the reduction in vegetation height and cover associated with trampling and defoliation can result in a harsher microclimate at the soil surface and diminished protection from predators. Furthermore, Piearce (1984) found no evidence to support the contention that higher numbers of dung pats in heavily trampled areas could counteract the effect of reduced vegetation and soil deformation. Regardless of the causal mechanism, reductions in earthworm numbers as a result of grazing could lead to a loss of the important beneficial activities carried out by the organisms, which includes recovery of the soil after compaction (Drewry, 2006).

VI. THE IMPACT OF DEFOLIATION BY GRAZING ANIMALS ON GRASSLAND VEGETATION

The vegetation of grasslands is central to the livestock/dairy production system. It provides forage for grazing animals and is often used as a source of food (in the form of hay or silage), while the animals are housed indoors over the winter period. Vegetation also protects the soil surface from the treading effects of grazing animals as well as the erosional influence of rainsplash and surface runoff. This, in turn, can help preserve the water quality in surface waters within these environments. However, the high stocking densities associated with intensively managed grasslands can lead to excessive defoliation of vegetation and can have serious implications on pasture herbage yield, vegetation percentage cover, and vegetation biodiversity. Livestock can defoliate large amounts of vegetation while grazing. The response of the grassland vegetation to this defoliation will depend on factors such as: (1) the frequency and severity of vegetation removal/grazing, (2) the degree of compaction, pugging, and poaching on the soil, and (3) the amount of excreta deposited onto the pasture. These factors are determined primarily by the species and age of grazing animal, the stocking density, soil texture, soil moisture content, and farm management factors, which are discussed below.

A. Animal Species and Age

Different species of animals graze differently in terms of how much they eat (quantity) and what they eat (selectivity). The quantity of vegetation consumed is a function of body mass and stage in the animal's life cycle. The selectivity is a consequence of differences in the animal's mouth size, lip anatomy, and method of prehension (Matches, 1992). Livestock are capable of consuming large amounts of vegetation while grazing. For example, estimates of the daily consumption by cows, based on UK studies, range from 7 kg dry matter per day for heifers (Rook *et al.*, 2004; Rutter *et al.*, 2002) to between 14 and 18 kg dry matter per day for dairy cows (Gibb *et al.*, 1999; Orr *et al.*, 2001; Rutter *et al.*, 2004). Given that dry matter constitutes around 20% of the fresh weight of vegetation, one dairy cow is capable of ingesting ~100 kg of fresh plant matter per day (Rook *et al.*, 2004). This is clearly a significant quantity of herbage consumption, particularly when the mean net rate of herbage growth is considered to be ~60 kg dry matter per hectare, per day for the United Kingdom intensively managed grasslands (Orr *et al.*, 1988). Furthermore, this vegetation removal is not likely to be evenly distributed across a designated grazing area. Grazing livestock display both positive selection of desirable areas (fresh young grass/clover shoots) and avoidance of undesirable areas (e.g., dung pats, coarse grasses) (Rook *et al.*, 2004), as well as tendencies to aggregate and spend a disproportionately large amount of time grazing and walking adjacent to fence lines. This can lead to the development of overgrazed patches, even to the extent of death of vegetation (Matches, 1992).

Selectivity of grazing by some animals can even be used to isolate preferred species of plant and individual parts of plants. For example, Hughes *et al.* (1984) found that the diet of lambs (consuming ~1.5 kg dry matter per day) contained a greater portion of clover (+23%) and a smaller portion of ryegrass (−19%) and of dead material (−3%) than that of calves (consuming ~7 kg dry matter per day), demonstrating some of the differences in grazing selectivity between species, and that lambs could be more selective while grazing than calves. The selectivity of grazing animals can be an important factor in determining the pasture species composition and the stability of the ecosystem.

B. Stocking Density

As stocking density is increased, the frequency and closeness of defoliation increases (Matches, 1992). This leads to lower herbage availability per animal which in turn can cause the animals to be less selective in what they eat (Matches, 1992). Therefore, as stocking rate increases, the level of plant defoliation increases along with changes in sward morphology and composition (Matches, 1992). The ultimate economic goal of intensive livestock farming is to maximize production at minimum expense. Pastoral production can be increased either by raising production per animal (by using, for example, concentrated feeds, heated housing, and veterinary medicines), or by increasing the stocking density (Langlands and Bennett, 1973). However, there are natural limits on production per animal imposed by the rates of natural internal biochemical processes. There are also natural limits to stocking density due to the rate of herbage growth/production. While herbage yield can be enhanced via the use of fertilizers and weed-control treatments, this enhancement cannot continue indefinitely. For decades, scientists have indeed considered stocking density in relation to the carrying capacity of the pasture vegetation (Evans, 1998). The carrying capacity of a pasture, in this case, is an estimate of how much and what type of vegetation will grow there, and in turn, how many grazing animals this vegetation can support (Evans, 1998; Matches, 1992). The optimum stocking density in terms of production is that at which the leaf area of vegetation is maintained at a level which allows for maximum growth rates throughout the grazing season. However, while the carrying-capacity concept considers the consumption rate of vegetation by grazing animals and the threat of overgrazing to vegetation yield, it does not consider the other environmental impacts of grazing animals or how these are affected by stocking density (Evans, 1998; Matches, 1992). Evans (1998) argues that the stocking density of a pasture will determine not only the total amount of vegetation consumed but also the total number of hooves impacting on the grassland soil surface, and the amount of excreta deposited onto the pasture, therefore, estimates of sustainable production should include these factors into stocking density planning. Overgrazing is the cause of 23% of the soil degradation in Europe (RCEP, 1996). Overgrazing is not a new phenomenon and there have been many cases of severe overgrazing in the United Kingdom in the past 200 years (Johns, 1998).

C. Vegetation Response

One of the most important impacts of defoliation is associated with the change in herbage yield. Changes to the productivity of grassland vegetation in response to grazing depend strongly on grazing intensity/stocking density. McNaughton (1983, 1986) and Hodgkinson and Mott (1986) have proposed three alternative hypotheses on how plant growth and fitness may respond to grazing intensity. Response one is where net primary productivity (NPP) of plants shows a consistent decline as the intensity of grazing increases. This is probably the most common view among ecologists and evolutionary biologists; it is based on the principle that herbivory is always detrimental to the plant eaten. Edmond (1958, 1962) reported this type of response from studies on perennial ryegrass and white clover pasture in New Zealand. The herbage yield from pasture (maintained in a wet state) with stocking densities of 3, 6, 9, 12, and 18 sheep per hectare were 808, 644, 490, 267, and 127 kg dry mass per 0.405 hectare, respectively (Edmond, 1962).

Response two is where the plants are able to compensate for tissue removal up to some level, beyond which plant productivity begins to decline as the intensity of defoliation increases further. Langlands and Bennett (1973) found this type of response from their five-year study on the effect of sheep stocking density on pastoral production in New South Wales, Australia. In this example, herbage production was relatively insensitive to increases in stocking density over the range of 2–22 sheep per hectare, but beyond this range herbage productivity began to decline at a greater rate with increasing stocking density. Langlands and Bennett (1973) attributed the decline in productivity at the higher stocking rates to the decline in basal cover and expansion of bare areas of soil.

Response three, perhaps the most interesting and controversial, is where moderate levels of defoliation may result in overcompensation by the plant, due to intrinsic or extrinsic consequences of defoliation (McNaughton, 1983). Thus, within some levels of defoliation, plant productivity may be enhanced. McNaughton (1983) termed this as "overcompensatory growth." Indeed, the principle that certain levels of defoliation may enhance and stimulate herbage growth has long been used in the production of turfgrasses (whereby frequent clipping at a moderate height is used) (Albert, 1927; Mortimer and Ahlgren, 1936). An example of this type of response was reported by Vickery (1972) who found that NPP was greatest at 20 sheep per hectare and least at 10 sheep per hectare, while NPP at 30 sheep per hectare was greater than at 10 sheep per hectare. Vickery (1972) attributed the lower NPP at 10 sheep per hectare to reduced photosynthesis as a result of canopy closure and increased plant competition in the absence of regular grazing. The highest NPP at 20 sheep per hectare was attributed to increased photosynthetic efficiency because of a higher proportion of younger tillers and plants at this stocking density (Vickery, 1972). The decline in NPP at the highest stocking density was attributed to the reduced plant material available for photosynthesis as a result of excessive herbage consumption.

McNaughton (1979) argued that simplified statements about the effect of damage to vegetative tissue on the ultimate yield of those or other tissues, implying a uniform monotonic series of effects on the plant by an increasing series of herbivory levels, are highly misleading. Rather the plant responds to a whole complex of environmental factors, and the variety of plant responses are subject to the constraints of plant genetics, developmental stage, and the plant tissues that are affected (McNaughton, 1983). Nevertheless, one key point which holds true in all of the above hypothetical responses is that at high levels of herbivory and defoliation plant productivity is negatively affected. Therefore, overstocking of grasslands may reduce livestock yield as well as having an impact on the plant system. If intensively managed grasslands are to be maintained in a productive yet environmentally friendly state, then a complete understanding of the effect of grazing on vegetation is critical. At present, however, a review of the literature reveals gaps in our knowledge regarding the responses of grassland vegetation to defoliation by grazing animals.

VII. IMPACT OF EXCRETION BY GRAZING ANIMALS ON VEGETATION, SOILS, AND SURFACE WATERS IN INTENSIVELY MANAGED GRASSLANDS

Livestock can produce large quantities of waste (urine and feces), with dairy cattle being the highest producers. For example, in the European Union, there are in excess of 24 million dairy cattle (Eurostat, 2006), each adult cow producing an average of ca. 20 tons of slurry (a mix of urine and feces) each year (Smith and Frost, 2000). On an annual basis, ~50% of this excreta is voided in the field while grazing (Chadwick and Chen, 2002), with the majority of the remainder being collected in the form of manure or slurry when the animals are housed indoors over the winter period (Mawdsley *et al*., 1995). The waste collected while the animals are housed indoors may also eventually be applied to the pasture surface through slurry spreading or manure application. This waste is a source of organic matter and nutrients [nitrogen (N) and phosphorus (P)] and is also a potential source of pathogens.

A. Livestock Wastes as a Source of Nutrients

Livestock wastes are often a rich source of nutrients such as N and P because only a small percentage (3–30%) of the nutrients in the food ingested by the animal is actually utilized by the animal and assimilated into its tissues, the remainder being excreted in feces and urine (Holmes, 1970; Tamminga, 1992). The nutrient content of livestock excreta may be enhanced further when the animals have been fed concentrated feeds (Tamminga, 1992). While the nutrient and organic matter content of animal waste can be considered to be beneficial to plant growth, long-term fertility, and soil structure in grassland environments, accumulation of nutrients in grassland soils has been shown to cause a shift in grassland plant diversity and botanical composition which has subsequent effects on the insect and invertebrate communities (Matches, 1992). Generally, the addition of N in urine stimulates growth of dominant grass species (such as *L. perenne*) and the addition of P in dung stimulates the growth of dominant legume species (such as *T. repens*) especially on P-deficient soil (Matches, 1992). In addition, plants immediately beneath dung pats may be killed due to absence of light, and urine occasionally scorches the sward (Matches, 1992). Furthermore, the accumulation of these nutrients in the soil and resultant increased delivery of excessive amounts of these nutrients (particularly N, P, and C) into surface waters is associated with eutrophication problems such as the growth of toxic algal blooms which pose a threat to the health of humans and domesticated and wild animals (Chadwick and Chen, 2002; Haygarth and Jarvis, 1999).

Defecation by grazing animals can also influence the distribution of nutrients in the soil and the spatial pattern of nutrients across the pasture. First, dung deposition on grassland tends to lead to higher concentrations of nutrients in the surface soil horizon in the absence of ploughing (Haygarth *et al.*, 1998). Second, there may be spatial excreta deposition hotspots associated with the movement behavior of livestock and the tendency to concentrate in camping-grounds or in sheltered spots overnight. These hotspots may be significant sources of readily available sediments and colloids, N and P, and pathogens, posing a high risk to surface water quality where these hotspots coincide with surface runoff flow paths (Page *et al.*, 2005), which may be promoted by the compaction and soil deformation effects of stock treading.

B. Livestock Wastes as a Source of Pathogens

The rumen and digestive tract of agricultural livestock is host to a rich diversity of microflora and can also act as a reservoir for pathogenic (disease-causing) microorganisms (Rasmussen *et al.*, 1993). Some of these pathogens are excreted in the feces of infected, and in some cases, healthy ‘‘carrier’’ animals (Chadwick and Chen, 2002). While some pathogens are obligate parasites and are of limited concern, others can survive saprophytically in the environment for long periods and pose a threat to other organisms (Mawdsley *et al.*, 1995). It has been suggested that modern intensive grassland management practices are contributing to greater abundance and survival of pathogens in livestock wastes. For example, prior to agricultural intensification, housed livestock were often bedded on large amounts of straw and the waste was managed as farmyard manure (Jones, 1982). Traditionally, this farmyard manure was composted, an aerobic process where temperatures often rise as high as 70 °C and therefore the majority of pathogens were destroyed (Jones, 1980). However, as herd size and the number of housed animals has increased, there has been a move toward the collection of waste in a semiliquid, slurry form which contains only a minimum amount of solid bedding material (Mawdsley *et al.*, 1995). It is estimated that 50–60% of waste from housed cattle is now managed as slurry (Smith and Unwin, 1983). However, in intensive systems, slurry is collected and stored under conditions which rapidly become anaerobic and hence temperature rise and the concurrent destruction of pathogens, seen in composting, does not occur (Rankin and Taylor, 1969). This slurry, containing pathogens, is often applied to the pasture surface where it potentially may be washed off into surface waters during rainfall events. The pathogens in livestock waste which pose the greatest threat to human health are bacterial pathogens such as *Eschericheria coli* O157 and *Salmonella* spp., viruses such as *Rotavirus* spp., and protozoa such as *Cryptosporidium* and *Gardia* spp. (Mawdsley *et al.*, 1995). These pathogens can be transferred to surface and drinking waters via hydrological transport in association with colloidal matter present in dung and soil (Chadwick and Chen, 2002).

VIII. IMPACTS OF GRAZING ANIMALS ON THE WATER QUALITY OF SURFACE WATERS IN INTENSIVELY MANAGED GRASSLANDS

The impacts of grazing animals on surface waters, unless livestock are allowed direct access to the channel network, are very much secondary effects resulting from the impacts of the grazing animals on the soils and vegetation of the grassland. This chapter has been structured around the principle that livestock carry out three key activities which may cause environmental degradation in intensively managed grassland environments: (1) treading, (2) defoliation, and (3) excretion. The mechanisms by which these activities can cause degradation of grassland soils and vegetation have already been discussed in previous sections; however, the ways in which this degradation can be transferred to surface waters have not yet been fully discussed. This section of the chapter examines the potential means by which grazing animals can indirectly impact on surface waters in grassland environments, as is illustrated in Fig. 8. This section is divided into the following parts: (A) soil erosion and sedimentation problems, (B) eutrophication, and (C) pathogenic

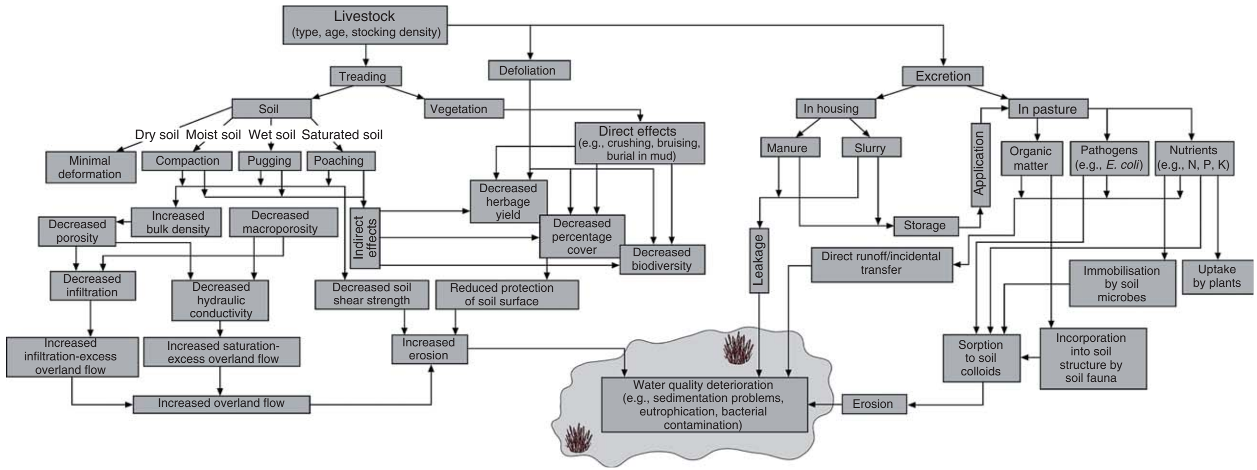

Figure 8 A conceptual model illustrating the impacts of treading, defoliation, and excretion by grazing animals on the water quality of surface waters in intensively managed grasslands.

contamination. These divisions reflect differences in the level of knowledge and understanding relating to each area.

A. Soil Erosion and Sedimentation Problems

Soil erosion can be defined as the removal of soil by wind, water, and mass movements at a faster rate than at which new soil forms (Morgan, 1980). While the rates of soil erosion in temperate regions are likely to be relatively small in comparison with those in tropical-humid regions, the impacts of soil erosion in temperate regions are certainly not insignificant when considering the environmental and economic costs which are incurred as a consequence of this process (Morgan, 1980). For example, the estimated cost of soil erosion to the UK economy is around £90 million per annum (Environment Agency, 2002). Furthermore, the costs of cleaning up the polluting impacts of soil erosion on water in the United Kingdom are estimated at £260 million per annum (Evans, 1995). The costs of soil erosion are therefore a result of both on-site and off-site impacts. On-site impacts are particularly important on agricultural land where they can lead to the redistribution of soil within a field, the loss of soil from a field, the breakdown of soil structure, and the decline in organic matter and nutrients. This, in turn, results in a reduction of cultivable soil depth and a decline in soil fertility (Morgan, 2005).

Off-site problems arise from the delivery of eroded sediment into surface waters, which reduces the capacity of rivers and drainage ditches, enhances the risk of flooding, blocks irrigation canals, and shortens the design life of reservoirs (Morgan, 2005; Verstraeten and Poesen, 2000). The cost of damages and dredging stream channels as a result of soil erosion from agriculture in the United Kingdom is estimated at £7.8 million per annum (Environment Agency, 2002). Sediment delivery to surface waters can also have direct ecological impacts by, for example, interfering with fish spawning/incubation sites (Greig *et al*., 2005; Walling *et al*., 2003). Salmon and trout in north-west Europe have declined from great abundance in preindustrial times to the present day, where they are absent from former habitats or where the threat of extinction to sparse residual populations is real (Harrod and Theurer, 2002). Causes of their reduction may be numerous and complex, but sediment intrusion into spawning gravels is one of them and has potential to cause serious damage to fish stocks (Greig *et al*., 2005; Harrod and Theurer, 2002; Walling *et al*., 2003). Sediment delivery to surface waters can also have indirect ecological impacts because these particles can act as vectors of sorbed contaminants such as pesticides (Morgan, 2005), pathogens (Oliver *et al*., 2005a) and P (Fraser *et al*., 1999; Sharpley *et al*., 1994; Svendsen *et al*., 1995). The sediment-facilitated transport of P to surface waters is a particularly significant threat to water quality because of the role that P plays in eutrophication.

While authors in the "grazing impacts" literature have often acknowledged and recognized that the changes to soil physical properties, vegetation cover, and pasture hydrology brought about by intensive grazing may have implications for increasing soil erosion, very little quantitative data exists to directly support this hypothesis from intensively managed grasslands. In fact, grasslands as a whole have largely been ignored as potential sources of sediments by the soil erosion community (Evans, 1998; Heathwaite *et al*., 1990; Heathwaite and Dils, 2000). In the United Kingdom particularly, the quantification of soil erosion has focused on the arable sector of the agricultural land (Boardman, 2002; Brazier *et al*., 2007; Evans, 1997; Morgan, 2005). This research bias has, in the past, been justified by the belief that grasslands do not yield significant amounts of sediment due to the effect of their high surface cover which acts to intercept raindrops and retard runoff, resulting in limited detachment and transport of soil particles (Nash and Halliwell, 1999; Nash and Murdoch, 1997). This chapter has highlighted several reasons why this perception may be false, particularly in intensively managed grasslands where (1) soil compaction, pugging, and poaching can promote surface runoff generation and reduce the resistance of the soil to erosion, (2) defoliation and direct and indirect treading effects can dramatically reduce the protective vegetation cover, and (3) feces deposited onto the pasture by grazing animals and slurry/manure applied to the pasture by farmers provide a readily available source of particulate material. However, no new research has proven these links as yet, so they remain largely anecdotal and theoretical. Evans (1997) recommends that further research is needed into the erosion initiated by grazing animals and recommends that national surveys of erosion by grazing animals should be conducted. This information is vital if governments are aiming to be able to effectively mitigate water quality issues in surface waters. Failure to recognize the potential sources of surface water pollutants in catchments, and continuation of a "perceptual understanding" of the factors that control the magnitude of these sources (rather than an empirical understanding), will inevitably result in failure of water-quality remediation measures.

B. Eutrophication

Eutrophication is a process of nutrient enrichment which increases the primary productivity of surface waters and can potentially impact on all water types ranging from those that are nutrient poor (oligotrophic) to those considered to be nutrient enriched (eutrophic) (Foy, 2005). Eutrophication is a slow and benign natural process associated with the ageing of a lake or waterbody; however, it can be accelerated and become harmful to ecosystem health if, for example, the anthropogenic input of nutrients occurs. The nutrients that are commonly of particular concern with regards to freshwater eutrophication are N and P. Phosphorus is most often the limiting nutrient in freshwater aquatic systems and is therefore commonly the prime cause of eutrophication (Sharpley and Rekolainen, 1997; Sharpley and Smith, 1990). The significance of P is evident from the strong correlation between mean total P (TP) measured in lakes over a wide geographic area and chlorophyll a, which is used as a surrogate for algal abundance (Canfield, 1983; Forsberg and Ryding, 1980; McCauley *et al*., 1989; OECD, 1982; Prairie *et al*., 1989; Pridmore *et al*., 1985; Seip *et al*., 2000; Smith, 1998). Even relatively low concentrations of P in surface waters can lead to eutrophication problems. The Organization for Economic Cooperation and Development (OECD) suggests that eutrophication problems can be triggered by P concentrations as low as 35–100 μg liter^{-1} (OECD, 1982).

For oligotrophic–mesotrophic waters, excessive P inputs can result in an increase in fish size which may be considered to be beneficial to some lake users, but in biodiversity terms can be damaging if other fish species are in decline (Foy, 2005). For eutrophic waters, the input of P may maintain or exacerbate a range of undesirable effects. Eutrophication of waters can cause problems with its use for fisheries, recreation, industry, and drinking. For example, P can promote the excessive growth of aquatic vegetation and algae. The senescence and decomposition of this matter can deplete water oxygen levels which may lead to the mortality of fish and other aquatic organisms (Heathwaite, 1994). In addition, the cyanobacteria or blue-green algae commonly associated with eutrophic waters present particular water-quality problems, as some species produce fast acting neurotoxins and slower acting hepatotoxins which can have serious adverse impacts on the health of humans and domesticated and wild animals (Foy, 2005). For example, at Rutland Water in Leicestershire (United Kingdom) in 1988, the bacteria *Clostridium botulinum*, which flourishes in anoxic sediments, developed to the extent that it caused botulism in birds and mammals (including domestic pets) using the lake (Heathwaite, 1994). In addition to toxins, cyanobacteria and some algae can produce other dissolved organic compounds (DOCs), principally geosmin and isoburneol, that can cause taste and odor problems (Cooke and Kennedy, 2001; Watson *et al*., 1999). If chlorine reacts with the DOCs formed by algal cell lysis or algal extraction during water treatment, potentially carcinogenic trihalomethanes can enter potable water supplies (Hoehn *et al*., 1980).

Phosphorus, as the most common limiting nutrient in freshwater ecosystems, plays a key role in determining the presence of such harmful substances (e.g., neuro- and hepatotoxins and DOCs) in surface waters. There are several ways in which intensively managed grasslands can contribute nutrients to surface waters, and thus contribute to eutrophication. (1) Runoff of dissolved and particulate nutrients found in animal excreta which was deposited on the pasture while the animal was grazing. (2) Runoff of dissolved and particulate nutrients found in animal excreta which was applied to the pasture surface in the form of slurry and/or manure (collected while the animals were housed indoors). (3) Runoff of dissolved nutrients found in fertilizers which were applied to the pasture to enhance the pasture herbage yield. (4) Erosion and delivery of nutrients that are sorbed to soil particles and colloids. Mechanisms (1), (2), and (3) have received a fair amount of research attention over the last decade (Chardon *et al*., 1997; Dougherty *et al*., 2004; Edwards and Withers, 1998; Foy, 2005; Hart *et al*., 2004; Haygarth and Jarvis, 1999; Haygarth *et al*., 1998; Heinonen-Tanski and Uusi-Kamppa, 2001; Hooda *et al*., 2001; Nash and Halliwell, 1999; Preedy *et al*., 2001). However, as discussed in Section VIII.A, the potential for mechanism (4) to operate in intensively managed grasslands has largely been overlooked.

C. Pathogenic Contamination

Impairment of waterways and receiving lakes by pathogenic pollution has a significant impact on human health and quality of life, with contamination of drinking water supplies and the closure of recreational surface waters being two common consequences (Jamieson *et al*., 2005). The pathogens which pose the greatest threat to human health are bacterial pathogens such as *E. coli* O157 and *Salmonella* spp., viruses such as *Rotavirus* spp., and protozoa such as *Cryptosporidium* and *Gardia* spp. All of these pathogens can often be found in livestock wastes (Mawdsley *et al*., 1995), consequently, livestock-based agriculture is one of the main nonhuman sources of this kind of water pollution (Vinten *et al*., 2004). The pathogens contained in livestock wastes may enter surface waters (1) directly by leakage of wastes held in buildings or stores to drainage systems, (2) indirectly following the application of waste to land, or (3) indirectly from feces deposited onto the pasture while the livestock are grazing (Aitken, 2003; Oliver *et al*., 2005b; Rodgers *et al*., 2003). However, despite the serious implications of this type of surface water pollution, many aspects of bacterial survival and transport are poorly understood (Jamieson *et al*., 2005). A review by Mawdsley *et al*. (1995) highlighted, in particular, the lack of direct information on the movement of pathogenic microorganisms present in livestock waste through the landscape to surface waters, although it is known that bacteria are often transported in association with fine sediment and colloids (suggesting that soil erosion may be an important mechanism involved in the transfer process).

IX. ENVIRONMENTAL DEGRADATION BY GRAZING ANIMALS: RECOVERY AND REMEDIATION

A. Natural Recovery of Soil Physical Condition Following Treading Damage

To a certain extent, some of the damage to soil physical condition induced by grazing animals can be reversed by natural processes. Natural recovery of soil physical properties has been shown to be cyclical in nature, associated with wetting and drying cycles, subsequent soil cracking, earthworm burrowing, root penetration and decay, and freeze–thaw cycles during the winter (Dexter, 1991; Drewry, 2006; Drewry and Paton, 2005; Greenland, 1981; Greenwood and McKenzie, 2001; Hodgson and Chan, 1984). The time taken for natural recovery of soil physical condition varies depending on soil type, extent of initial damage, management methods, and climate, but may take anything from weeks to months, or even years (Drewry, 2006). A study by Drewry *et al*. (2004) found that most of the soil damage that occurred on a dairy farm in New Zealand took place in the wet spring and recovery of the soils physical condition occurred over the summer and autumn months, while recovery in the winter was much lower. The potential for natural recovery of soil physical condition needs to be balanced with the potential for further soil physical deterioration when regrazed and so relies heavily on land management factors (Drewry, 2006).

B. Mitigation and Damage Reduction Methods

Damage to the soil by grazing animals can never be entirely avoided, but the damage can be minimized by intelligent land and grazing management (Kellett, 1978). This section of the chapter focuses on mitigation and damage reduction measures for reducing the impact of grazing animals. In this chapter, the measures have been divided into three categories: (1) livestock management, (2) land management, and (3) waste management. Examples of some of the methods from each category are discussed below and can be found in Table I.

1. Livestock Management

Perhaps one of the most obvious methods for reducing the amount of damage to grassland vegetation and soil physical condition is that of reducing the total stocking density. Many authors suggest this as a remediation strategy (Kellett, 1978; Langlands and Bennett, 1973; Mulholland and Fullen, 1991;

Table I
Remediation and Mitigation Measures for Minimizing the Impacts of Grazing Animals

Method	References
Reduce stocking density	Patto *et al.* (1978)
Move livestock into housing or hard standings when the soil is wet or saturated	Kellett (1978)
Move grazing animals to drier areas of the pasture during wet periods	Sears (1956)
Move grazing animals to sacrifice enclosures within the pasture when the soil is wet or saturated	Mulholland and Fullen (1991)
Increase hoof contact area through the use of livestock shoes	Wind and Schothorst (1964)
Reduce the length of the grazing season	Davies and Armstrong (1986)
Reduce dietary N and P intake from animal feeds	Tamminga (1992)
Relocate feeding and drinking troughs in the pasture at regular intervals	Hilton (2002)
Install subsurface drainage	Armstrong and Garwood (1991); Davies and Armstrong (1986)
Loosen compacted soil layers in grassland fields	Harrison *et al.* (1994)
Terrace slopes	Gassman *et al.* (2006)
Tillage and reseed	Johnson *et al.* (1993)
Increase storage capacity of slurry and manure stores	McGechan and Wu (1998)
Switch from slurry to manure handling	Mawdsley *et al.* (1995)
Adopt a batch storage method for slurry and manure storage	Chadwick and Chen (2002)
Avoid applying slurry onto high risk areas and at high risk times	Haygarth and Jarvis (1999)
Inject slurry into the soil rather than spreading it with a splash plate	Hilton (2002)
Integrate fertilizer and manure/slurry nutrient supply	Unwin *et al.* (1986)

Patto *et al.*, 1978; Willatt and Pullar, 1983). This measure works in a number of ways. First, it reduces the number of hooves and the frequency that hooves impact on the pasture surface which in turn reduces the amount of soil deformation (Patto *et al.*, 1978), and damage to vegetation (Di *et al.*, 2001; Kellett, 1978). Second, it reduces the frequency and closeness of defoliation (Matches, 1992), which in turn reduces the occurrence of bare patches and promotes the development of a healthy vegetation cover which acts to protect the soil surface (Evans, 1997). Third, it decreases both the amount of excreta deposited onto the pasture while grazing and the amount of excreta collected while the animals are housed indoors (which may be spread onto the field at a later date). This in turn lowers the potential for N, P, and pathogens (found in animal waste) to be transported from land to surface waters. However, while this measure would be relatively simple to implement, it would result in a reduction in farm income which could threaten the economic viability of production.

A less drastic change for livestock management would be to move livestock into housing or onto hard standings when the grassland soil is wet or saturated (Kellett, 1978). This method is based on the principle that the resistance of a soil to deformation under treading declines as soil moisture increases and therefore the greatest amount of soil damage occurs when livestock tread on wet soils (Climo and Richardson, 1984; Patto *et al.*, 1978; Wind and Schothorst, 1964). If livestock are removed from the pasture during these high risk times, damage to soils and vegetation will be limited. Farmers can maintain a regular assessment of the weather forecast and act accordingly. Preventing treading while rain is occurring and water is ponding on the soil surface has been suggested as a very simple but particularly effective method for reducing poaching damage (Scholefield and Hall, 1985). However, this dynamic form of management is not always possible as housing and hard standings may already be in use or may not even exist on some farms. Perhaps a more practical alternative to this would be to move livestock to drier areas (if present) of the pasture during wet periods (Sears, 1956), or to move livestock into sacrifice enclosures of the pasture where they are allowed to damage only a small area (Mulholland and Fullen, 1991). With the latter method, the farmer would need to ensure that these sacrifice areas are poorly connected to the channel network to prevent eroded sediment and colloidal material from entering surface waters.

Another measure which works by the same mechanism as above is that of reducing the length of the grazing season. In many temperate countries, precipitation is seasonal with the grazing season fitting between two hydrological seasons. However, with this livestock management method, there is a risk of animals being on the pasture while it is still wet (spring), or becoming wet (autumn). By reducing the length of the grazing season, the chance of livestock treading on wet soils is reduced (Davies and Armstrong, 1986). One issue with this damage reduction method is that it requires a greater amount of stored food (silage, hay, concentrated feeds) while the animals are housed indoors. These come at a cost to the farmer and can result in increased external N and P being brought into the grassland system.

A novel suggestion by Wind and Schothorst (1964) involved the use of a type of shoe for livestock. This method is based on the principle that the forces imposed on the soil (which cause compaction, pugging, and poaching) by animal hooves are a function of animal mass and the surface area of the hooves in contact with the ground. By increasing the surface area of animal hooves (through the use of a type of shoe), the forces imposed on the soil are reduced and less deformation will take place as a result. Wind and Schothorst (1964) propose that relatively strong shoes could be made in bulk for little expense. However, while this suggestion makes good logical sense, there is little evidence to show that the shoes would actually be successful with real livestock.

A method for reducing N and P contents of animal excreta and therefore N and P accumulation in soils and runoff to surface waters is that of reducing dietary N and P intake. In modern intensively managed grasslands, the diet of livestock is often supplemented with concentrated feeds which are enriched with N and P in excess of the animals' requirements. This maintains high yields from the animals; however, any N and P in excess of animal requirements is excreted by the animal (Holmes, 1970; Tamminga, 1992). By reducing the N and P content of livestock feeds, this reduces N and P concentrations in excreta, reducing N and P accumulation in grassland soils and reducing the potential for N and P transfer to surface waters where they can contribute to eutrophication problems.

2. Land Management

One large-scale and long-term land management strategy for the reduction of damage to soil physical condition by grazing animals is the installation of subsurface drainage (Armstrong and Garwood, 1991; Davies and Armstrong, 1986; Kellett, 1978; Mulholland and Fullen, 1991). The installation of subsurface drainage leads to a lowering of the water table level and a decrease in the moisture content of the surface soil, thus increasing the shear strength of the soil and its resistance to damage (Patto *et al*., 1978). Several workers have reported that soil damage is reduced when the water table is lowered from the surface, and is normally avoided where the water table is kept below 500 mm of the surface (Davies and Armstrong, 1986; Kellett, 1978; Patto *et al*., 1978). However, calculations of the pipe drain spacing required for an effective drainage scheme on a typical grassland site in the United Kingdom, for example, reveal that pipes would need to be too close to be economically justified (Davies and Armstrong, 1986; Kellett, 1978). A less costly method of achieving this desired drainage involves the use of mole drainage over wide-spaced lateral pipes. This has been traditionally used in only a few grassland areas (Armstrong and Garwood, 1991), but where it has been used, it has been very successful at improving water removal from the surface soil and reducing damage to the soil by grazing animals (Davies and Armstrong, 1986; Kellett, 1978). However, while pipe and mole drainage has been proven to control soil damage on soils with a high clay content (Davies and Armstrong, 1986), it is not appropriate for soils with a clay content of $<30\%$ because of instability and collapse of the channels. Furthermore, there is uncertainty over how the installation of subsurface drainage influences sediment and nutrient transfers from land to surface waters and there is evidence to suggest that subsurface drains may, in some cases, act as preferential pathways for runoff and provide a direct conduit to watercourses (Chapman *et al*., 2001; Dils and Heathwaite, 1999; Heathwaite *et al*., 2005; Simard *et al*., 2000).

A shorter-term land management method for mitigating soil and vegetation damage, such as compaction, pugging, and poaching, involves tillage and reseeding (Johnson *et al.*, 1993). This process breaks up layers of compacted soil and removes deep hoof prints, allowing vegetation to reestablish. However, a review of the literature on the effects of tillage by Greenwood and McKenzie (2001) revealed variable responses. Dexter (1991) suggested that although a compacted grassland soil can be temporarily turned into a soil with apparently near-perfect structure by tillage (e.g., a seed bed of 1- to 5-mm diameter aggregates overlying a loosened, well-drained subsoil), the structure produced in this way may be far from equilibrium, may be mechanically unstable, and may collapse when wet to be as bad, if not worse, as before tillage. Furthermore, tillage can accelerate the deterioration of the soil physical condition by, for example, accelerating decomposition of organic matter and by disrupting stable soil aggregates (Dexter, 1991). Tillage and reseeding is probably best used over small areas such as old sites of drinking troughs/feeding troughs and gateways which have been moved, where the soil requires loosening and rejuvenating (Harrison *et al.*, 1994). If tillage and reseeding is used over whole fields or significantly large areas, there is a risk of enhanced erosion if rainfall occurs shortly after tillage and before the vegetation has established.

An alternative, more dynamic form of land management involves the regular movement of drinking and feeding troughs. The soil and vegetation around these features tend to receive the greatest damage by grazing animals because animals tend to congregate around them and so they are exposed to more frequent treading, defoliation, and defecation and so eventually become compacted, pugged, or poached, devoid of vegetation, and rich in excreta. These areas can then become critical source areas (CSAs) for sediment, N, P, and pathogens, threatening water quality in surface waters. Moving the troughs before significant visible degradation occurs can reduce pasture damage and minimize environmental consequences (Hilton, 2002). Similarly, gateways also receive a higher frequency of animal traffic than the rest of the pasture and so can become degraded areas and eventually CSAs for surface waters. Farmers need to make sure that these features are located away from the channel network. Alternatively, connectivity to the channel network can be reduced through the use of landscape features such as buffer strips and/or hedges (Hilton, 2002).

3. Waste Management

As mentioned previously, livestock can produce large quantities of waste (urine and feces). Approximately 50% of this waste is collected while the animals are housed indoors and is stored in the form of slurry (a liquid mix of urine and feces) or manure (a heap of solid dung and bedding material). This waste can be recycled to land at some point and if used efficiently and effectively can provide a host of benefits to both farmers and the environment (Oliver *et al*., 2005a). For example, recycling livestock waste to land prevents excessive fecal waste accumulation in farm storage systems (Unwin *et al*., 1986), builds soil quality, and returns valuable nutrients back to the grassland system (Hooda *et al*., 2000). However, if poorly managed, livestock wastes can present a threat to environmental quality if, for example, farm storage systems leak or overspill into streams or if runoff from grassland contains recently applied animal wastes which enter surface waters. One major factor contributing to poor waste management is related to the storage capacity of farm waste facilities. For example, farmers are often under pressure to apply slurry to in unsuitable conditions during the winter period, due to limited farm storage facilities (Chadwick and Chen, 2002; Edwards and Withers, 1998). There is much anecdotal evidence of slurry applications that occur in defiance of good agricultural practice, including slurry spreading onto wet or frozen land, and the ejection of slurry from roadways and farm tracks onto hillslopes of adjacent fields (Preedy *et al*., 2001). These practices are likely to promote pollution of surface waters. A simple remediation measure for this issue would be to increase the capacity of farm waste stores (McGechan and Wu, 1998). This would relieve pressure on farmers to empty their stores and apply waste in unsuitable conditions. Clearly, the enlargement of stores will come at a cost to the farmer. However, costs can be minimized if the farmer shifts to collecting waste in the form of solid manure rather than slurry, as it will not always be necessary to construct purpose built stores for manure handling. Solid manure can be stored temporarily in heaps on hard-standings or at suitable locations within the farm and therefore the storage capacity for manure is inherently more flexible than that used for slurry (a fixed volume pit, tank, or reservoir).

Another advantage of solid manure handling, as opposed to slurry collection, is associated with the reduced survivorship and abundance of pathogens (Jones, 1982; Mawdsley *et al*., 1995). As mentioned previously, the aerobic composting process can produce temperatures of up to 70°C which can kill off many pathogens (Jones, 1982). In contrast, slurry storage leads to anaerobic conditions and therefore temperatures are not raised high enough to kill off as many pathogens. An alternative method for reducing the number of pathogens in stored animal wastes is that of adopting a batch storage method for both slurry and manure storage, whereby only the older waste is applied to the land, while the fresher waste is being collected in a different batch until it has decayed/composted further. This decreases the number of pathogens applied to the land due to the pathogen die-off process during storage (Oliver *et al*., 2005a). A further improvement for reducing runoff of applied animal wastes from land to surface waters involves a method of injecting the slurry into the soil rather than spreading it over the surface (Heinonen-Tanski and Uusi-Kamppa, 2001). Heinonen-Tanski and Uusi-Kamppa (2001) found that injecting the slurry rather than spreading it on the surface, reduced total P and total N in runoff from grassland by an average of 81% and 73%, respectively.

X. FUTURE RESEARCH

One of the defining characteristics of intensively managed grassland is the high stocking density. Authors often cite the stocking density of a pasture to be one of the most important factors controlling the magnitude and extent of environmental degradation. The stocking density determines the number of hooves and frequency of hooves impacting on the soil surface, amount, frequency and closeness of defoliation, and the amount of excreta deposited onto the pasture. At low stocking densities, grazing can be beneficial to the environment, enhancing nutrient cycling and promoting biodiversity. However, at high stocking densities, damage to the pasture may occur, threatening the sustainability of farming and potentially impacting on water quality in surface waters. One simple remediation measure for the reduction of this environmental degradation would therefore be to reduce livestock stocking densities to a more optimum level whereby environmental degradation is limited but economic viability is maintained. The question is, "what is this optimum stocking density?" At present, this question cannot be satisfactorily answered due to the lack of research in this area. Nevertheless, evidence suggests that the optimum stocking density in terms of minimal environmental degradation is likely to vary between environments, depending on factors such as soil texture, topography, and the presence/absence of subsurface drainage. It is also likely to vary over time, fluctuating both seasonally (taking into account seasonality of plant growth, soil moisture content) and annually (taking into account climatic fluctuations which influence plant growth and soil moisture). It is also likely to be scale-dependent due to animal behavioral patterns and the tendency for animals to congregate in certain areas of the field. For example, a stocking density of 2 LSU per hectare may cause little damage in a 1-ha field. However, the same stocking density in a 30 ha field may result in much higher levels of damage to the pasture because the livestock tend to concentrate at certain points in the field (drinking/feeding troughs, fence-lines, gateways, sheltered spots) and therefore the effective stocking density will be exaggerated at these points. There may be up to 60 LSU in less than 5% of the total area (in the above example). In the current market, the optimum stocking density for minimal environmental degradation may not be economically viable for many small conventional farmers. However, this is partly a problem associated with the value of produce and the amount that retailers and consumers are prepared to pay for environmentally and economically sustainable farming. At present, supermarket "price wars" on produce, such as milk and meat, threatens both the future of farming and the environment. Future research needs to quantify the link between damage to grassland soils/vegetation by grazing animals and the rate of soil erosion and sediment delivery to surface waters. Furthermore, there is a demand for scientists to determine the most sustainable stocking density for grasslands with differing environmental characteristics. Clearly, this will need to consider the socioeconomic aspects of farming as well as the environmental consequences.

ACKNOWLEDGMENTS

The authors are grateful to David Scholefield and Les Firbank of IGER North Wyke for comments on the chapter. We acknowledge UK Defra Projects PE0120 and PE0118 for funding that resulted in this work. IGER and SoilCIP are supported by core funding from the UK Biotechnology and Biological Sciences Research Council (BBSRC).

References

Abdel-Magid, A. H., Trlica, M. J., and Hart, R. H. (1987). Soil and vegetation responses to simulated trampling. *J. Range Manage.* **40**, 303–306.

Aitken, M. N. (2003). Impact of agricultural practices and river catchment characteristics on river and bathing water quality. *Water Sci. Technol.* **48**, 217–224.

Albert, W. B. (1927). Studies on the growth of alfafa and some perennial grasses. *J. Am. Soc. Agron.* **19**, 624–654.

Alderfer, R. B., and Robinson, R. R. (1947). Runoff from pastures in relation to grazing intensity and soil compaction. *J. Am. Soc. Agron.* **39**, 948–958.

Alexandrou, A., and Earl, R. (1997). Development of a technique for assessing the behaviour of soil under load. *J. Agric. Eng. Res.* **68**, 169–180.

Armstrong, A. C., and Garwood, E. A. (1991). Hydrologic consequences of artificial drainage of grassland. *Hydrol. Process.* **5**, 157–174.

Boardman, J. (2002). The need for soil conservation in Britain-revisited. *Area* **34.3**, 419–427.

Brazier, R. E., Bilotta, G. S., and Haygarth, P. M. (2007). A perspective on the role of lowland, agricultural grasslands in contributing to erosion and water quality problems in the UK. *Earth Surface Processes and Landforms* **36** (in press).

Bryant, H. T., Blaser, R. E., and Peterson, J. R. (1972). Effect of trampling by cattle on bluegrass yield and soil compaction of a Meadowville loam. *Agron. J.* **64**, 331–334.

Canfield, D. E. (1983). Prediction of chlorophyll a concentration in Florida lakes: The importance of phosphorus and nitrogen. *Water Resour. Bull.* **19**, 255–262.

Carroll, Z. L., Reynolds, B., Emmett, B. A., Sinclair, F. L., Ruiz de Ona, C., and Williams, P. (2004). "The Effect of Stocking Density on Soil in Upland Wales." Rep. No. Countryside

Council for Wales Contract Science Report Number 630. centre for Ecology and Hydrology, Bangor.
Carter, J. F. (1962). "Pastures and Range Research Techniques." Comstock Publishing Associates, NY.
Chadwick, D. R., and Chen, S. (2002). Manures. *In* "Agriculture, Hydrology and Water Quality" (P. M. Haygarth and S. C. Jarvis, Eds.), pp. 437–451. CAB International, Wallingford.
Chapman, A. S., Foster, I. D. L., Lees, J. A., Hodgkinson, R. A., and Jackson, R. H. (2001). Particulate phosphorus transport by subsurface drainage from agricultural land in the UK. Environmental significance at the catchment and national scale. *Sci. Total Environ.* **266**, 95–102.
Chardon, W. J., Oenema, O., del Castilho, P., Vriesema, R., Japenga, J., and Blaauw, D. (1997). Organic phosphorus solutions and leachates from soils treated with animal slurries. *J. Environ. Qual.* **26**, 372–378.
Climo, W. J., and Richardson, M. A. (1984). Factors affecting the susceptibility of 3 soils in the Manawatu to stock treading. *N. Z. J. Agric. Res.* **27**, 247–253.
Cluzeau, D., Binet, F., Vertes, F., Simon, J. C., Riviere, J. M., and Trehen, P. (1992). Effects of intensive cattle trampling on soil-plant-earthworms system in two grassland types. *Soil Biol. Biochem.* **24**, 1661–1665.
Cooke, G. D., and Kennedy, R. H. (2001). Managing drinking water supplies. *Lakes Reserv. Manage.* **17**, 157–174.
Davies, P. A., and Armstrong, A. (1986). Field measurements of grassland poaching. *J. Agric. Sci.* **106**, 67–73.
Defra (2005). "Agriculture in the United Kingdom." HMSO, London.
Dexter, A. R. (1991). Amelioration of soil by natural processes. *Soil Tillage Res.* **20**, 87–100.
Di, H. J., Cameron, K. C., Milne, J., Drewry, J. J., Smith, N. P., Hendry, T., Moore, S., and Reijnen, B. (2001). A mechanical hoof for simulating animal treading under controlled conditions. *N. Z. J. Agric. Res.* **44**, 111–116.
Dils, R. M., and Heathwaite, A. L. (1999). The controversial role of tile drainage in phosphorus export from agricultural land. *Water Sci. Technol.* **39**, 55–61.
Dougherty, W. J., Fleming, N. K., Cox, J. W., and Chittleborough, D. J. (2004). Phosphorus transfer in surface runoff from intensive pasture systems at various scales: A review. *J. Environ. Qual.* **33**, 1973–1988.
Drewry, J. J. (2006). Natural recovery of soil physical properties from treading damage of pastoral soils in New Zealand and Australia: A review. *Agric. Ecosyst. Environ.* **114**, 159–169.
Drewry, J. J., and Paton, R. J. (2005). Effects of sheep treading on soil physical properties and pasture yield of newly sown pastures. *N. Z. J. Agric. Res.* **48**, 39–46.
Drewry, J. J., Cameron, K. C., and Buchan, G. D. (2001). Effects of simulated dairy cow treading on soil physical properties and ryegrass pasture yield. *N. Z. J. Agric. Res.* **44**, 181–190.
Drewry, J. J., Paton, R. J., and Monaghan, R. M. (2004). Soil compaction and recovery cycle on a Southland dairy farm: Implications for soil monitoring. *Aust. J. Soil Res.* **42**, 851–856.
Edmond, D. B. (1958). The influence of treading on pasture. A preliminary study. *N. Z. J. Agric. Res.* **3**, 319–328.
Edmond, D. B. (1962). Effects of treading perennial ryegrass (*Lolium perenne* L.) and white clover (*Trifolium repens* L.) pastures in winter and summer at two soil moisture levels. *N. Z. J. Agric. Res.* **6**, 265–276.
Edwards, A. C., and Withers, P. J. A. (1998). Soil phosphorus management and water quality: A UK perspective. *Soil Use Manage.* **14**, 124–130.
Environment Agency (2002). "Agriculture and Natural Resources: Benefits, Costs, and Potential Solutions." Environment Agency, Bristol.

Eurostat (2006). EUROSTAT. http://ec.europa.eu/comm/dgs/eurostat/index.en.htm (online resource, accessed 18 August 2006) Brussels.

Evans, R. (1995). Soil erosion and land use: Towards a sustainable policy. *In* "Proceedings of the Seventh Professional Environmental Seminar" (R. Evans, Ed.), pp. 14–26. White Horse Press, Cambridge.

Evans, R. (1997). Soil erosion in the UK initiated by grazing animals. A need for a national survey. *Appl. Geogr.* **17**, 127–141.

Evans, R. (1998). The erosional impacts of grazing animals. *Prog. Phys. Geogr.* **22**, 251–268.

Federer, C. A., Tenpes, G. H., Schmidt, D. R., and Tanner, C. B. (1961). Pasture compaction by animal traffic. *Agron. J.* **53**, 53–54.

Forsberg, C., and Ryding, S. O. (1980). Eutrophication parameters and trophic state indices in 30 Swedish waste-receiving lakes. *Archiv. fur Hydrobiol.* **88**, 189–207.

Foy, R. H. (2005). The return of the phosphorus paradigm: Agricultural phosphorus and eutrophication. *In* "Phosphorus: Agriculture and the Environment" (J. T. Sims and A. N. Sharpley, Eds.), Vol. Agronomy Monograph No. 46, pp. 1121. Madison, Wisconsin.

Frame, J. (1971). Fundamentals of grassland management. Pt. 10. The grazing animal. *Scot. Agric.* **50**, 28–44.

Fraser, A. I., Harrod, T. R., and Haygarth, P. M. (1999). The effect of rainfall intensity on soil erosion and particulate phosphorus transfer from arable soils. *Water Sci. Technol.* **39**, 41–45.

Gassman, P. W., Osei, E., Saleh, A., Rodecap, J., Nowell, S., and Williams, J. (2006). Alternative practices for sediment and nutrient loss control on livestock farms in northeast Iowa. *Agric. Ecosyst. Environ.* **117**, 135–144.

Gibb, M. J., Huckle, C. A., Nuthall, R., and Rook, A. J. (1999). The effect of physiological state (lactating or dry) and sward surface height on grazing behaviour and intake by dairy cows. *Appl. Anim. Behav. Sci.* **63**, 269–287.

Greenland, D. J. (1981). Soil management and soil degradation. *J. Soil Sci.* **32**, 301–322.

Greenwood, K. L., and McKenzie, B. M. (2001). Grazing effects on soil physical properties and the consequences for pastures: A review. *Aust. J. Exp. Agric.* **41**, 1231–1250.

Greenwood, K. L., Macleod, D. A., and Hutchinson, K. J. (1997). Long-term stocking rate effects on soil physical properties. *Aust. J. Exp. Agric.* **37**, 413–419.

Greig, S. M., Sear, D. A., and Carling, P. A. (2005). The impact of fine sediment accumulation on the survival of incubating salmon progeny: Implications for sediment management. *Sci. Total Environ.* **344**, 241–258.

Harris, W. P. (1971). Compaction of agricultural soils—The soil compaction process. *Am. Soc. Agric. Eng.*

Harrison, D. F., Cameron, K. C., and Mclaren, R. G. (1994). Effect of subsoil loosening on soil physical properties, plant root growth and pasture yield. *N. Z. J. Agric. Res.* **37**, 559–567.

Harrod, T. R., and Theurer, F. D. (2002). Sediment. *In* "Agriculture, Hydrology and Water Quality" (P. M. Haygarth and S. C. Jarvis, Eds.), p. 502. CABI, Wallingford.

Hart, M., Quin, B. F., and Nguyen, M. L. (2004). Phosphorus runoff from agricultural land and direct fertilizer effects: A Review. *J. Environ. Qual.* **33**, 1954–1972.

Haygarth, P. M., and Jarvis, S. C. (1999). Transfer of phosphorus from agricultural soils. *Adv. Agron.* **66**, 196–249.

Haygarth, P. M., Chapman, P. J., Jarvis, S. C., and Smith, R. V. (1998). Phosphorus budgets for two contrasting grassland farming systems in the UK. *Soil Use Manage.* **14**, 160–167.

Heathwaite, A. L., and Dils, R. M. (2000). Characterising phosphorus loss in surface and subsurface hydrological pathways. *Sci. Total Environ.* **251–252**, 523–538.

Heathwaite, A. L., and Johnes, P. J. (1996). The contribution of nitrogen species and phosphorus fractions to stream water quality in agricultural catchments. *Hydrol. Process.* **10**, 971–983.

Heathwaite, A. L., Burt, T. P., and Trudgill, S. T. (1990). Land-use controls on sediment production in a lowland catchment, south-west England. *In* "Soil Erosion on Agricultural Land" (J. Boardman, I. D. L. Foster, and J. A. Dearing, Eds.), pp. 70–86. John Wiley and Sons Ltd., Chichester.

Heathwaite, L. (1994). Eutrophication. *Geogr. Rev.* **7**, 31–37.

Heathwaite, L., Haygarth, P., Matthews, R., Preedy, N., and Butler, P. (2005). Evaluating colloidal phosphorus delivery to surface waters from diffuse agricultural sources. *J. Environ. Qual.* **34**, 287–298.

Heinonen-Tanski, H., and Uusi-Kamppa, J. (2001). Runoff of faecal microorganisms and nutrients from perennial grass ley after application of slurry and mineral fertiliser. *Water Sci. Technol.* **43**, 143–146.

Hillel, D. (1980). "Fundamentals of Soil Physics." Academic Press, London.

Hilton, J. (2002). "Reducing Diffuse Pollution to Rivers: A Dictionary of Best Management Practices." English Nature, Peterborough.

Hodgkinson, K. C., and Mott, J. J. (1986). On coping with grazing. *In* "Grazing-Lands Research at the Plant-Animal Interface" (F. P. Horn, Ed.). Winrock International, Morrilton.

Hodgson, A. S., and Chan, K. Y. (1984). Deep moisture extraction and crack formation by wheat and safflower in a vertisol following irrigated cotton rotations. *Rev. Rural Sci.* **5**, 299–304.

Hoehn, R. C., Barnes, D. B., Thompson, B. C., Randall, C. W., Grizzard, T. J., and Shaffer, P. T. B. (1980). Algae as a source of trihalomethane precursors. *Am. Water Works Assoc. J.* **72**, 344–349.

Holmes, W. (1970). Animals for food. *Proc. Nutr. Soc.* **29**, 237–244.

Hooda, P. S., Edwards, A. C., Anderson, H. A., and Miller, A. (2000). A review of water quality concerns in livestock farming areas. *Sci. Total Environ.* **250**, 143–167.

Hooda, P. S., Truesdale, V. W., Edwards, A. C., Withers, P. J. A., Aitken, M. N., Miller, A., and Rendell, A. R. (2001). Manuring effects in phosphorus accumulation in soils and potential environmental implications. *Adv. Environ. Res.* **5**, 13–21.

Hughes, T. P., Sykes, A. R., and Poopi, D. P. (1984). Diet selection of young ruminants in late spring. *Proc. N. Z. Soc. Anim. Prod.* **44**, 109–112.

Isselstein, J., Jeangros, B., and Pavlu, V. (2005). Agronomic aspects of biodiversity targeted management of temperate grasslands in Europe—A review. *Agron. Res.* **3**, 139–151.

Jamieson, R. C., Joy, D. M., Lee, H., Kostaschuk, R., and Gordon, R. J. (2005). Re-suspension of sediment-associated *Esherichia coli* in a natural stream. *J. Environ. Qual.* **34**, 581–589.

Johns, M. (1998). The impact of grazing and upland management on erosion and runoff. Environment Agency Technical Report, p. 123. Environment Agency, Bristol.

Johnson, R. J., McCallum, D. A., and Thomson, N. A. (1993). Pasture renovation after winter pugging damage. *Proc. N. Z. Grassland Assoc.* **55**, 143–146.

Jones, P. W. (1980). Disease hazards associated with slurry disposal. *Br. Vet. J.* **136**, 529–541.

Jones, P. W. (1982). Waste and animal health. *Public Health Eng.* **10**, 35–39.

Kellett, A. J. (1978). "Poaching of Grassland and the Role of Drainage. Technical Report 78/1." Ministry of Agriculture, Fisheries and Food, Cambridge.

Knight, D. (1992). The role of earthworms in managed permanent pastures in Devon, England. *Soil Biol. Biochem.* **24**, 1511–1517.

Kurz, I., O'Reilly, C. D., and Tunney, H. (2006). Impact of cattle on soil physical properties and nutrient concentrations in overland flow from pasture in Ireland. *Agric. Ecosyst. Environ.* **113**, 378–390.

Langlands, J. P., and Bennett, I. L. (1973). Stocking intensity and pastoral production. 1. Changes in the soil and vegetation of a sown pasture grazed by sheep at different stocking rates. *J. Agric. Sci.* **81**, 193–204.

Matches, A. G. (1992). Plant response to grazing: A review. *J. Prod. Agric.* **5**, 1–7.
Mawdsley, J. L., Bardgett, R. D., Merry, R. J., Pain, B. F., and Theodoru, M. K. (1995). Pathogens in livestock waste; their potential for movement through soil and environmental pollution. *Appl. Soil Ecol.* **2**, 1–15.
McCauley, E., Downing, J. A., and Watson, S. (1989). Sigmoid relationships between nutrient and chlorophyll among lakes. *Can. J. Fish. Aquat. Sci.* **46**, 1171–1175.
McDowell, R. W., Drewry, J. J., Muirhead, R. W., and Paton, R. J. (2003). Cattle treading and phosphorus and sediment losses in overland flow from grazed cropland. *Aust. J. Soil Res.* **41**, 1521–1532.
McGechan, M. B., and Wu, L. (1998). Environmental and economic implications of some slurry management options. *J. Agric. Eng. Res.* **71**, 273–283.
McNaughton, S. J. (1979). Grazing as an optimisation process: Grass-ungulate relationships in the Serengeti. *Am. Nat.* **113**, 691–703.
McNaughton, S. J. (1983). Compensatory plant growth as a response to herbivory. *Oikos* **40**, 329–336.
McNaughton, S. J. (1986). On plants and herbivores. *Am. Nat.* **128**, 765–770.
Monaghan, R. M., Paton, R. J., Smith, L. C., Drewry, J. J., and Littlejohn, R. P. (2005). The impacts of nitrogen fertilisation and increased stocking rate on pasture yield, soil physical condition and nutrient losses in drainage from a cattle-grazed pasture. *N. Z. J. Agric. Res.* **48**, 227–240.
Morgan, R. P. C. (1980). Soil erosion and conservation in Britain. *Prog. Phys. Geogr.* **4**, 24–47.
Morgan, R. P. C. (2005). "Soil Erosion and Conservation," 3rd ed. Blackwell Publishing, Oxford.
Mortimer, G. B., and Ahlgren, H. L. (1936). Influence of fertilisation, irrigation, and stage and height of cutting on yield and composition of Kentucky blue-grass. *J. Am. Soc. Agron.* **28**, 515–533.
Mulholland, B., and Fullen, M. A. (1991). Cattle trampling and soil compaction on loamy sands. *Soil Use Manage.* **7**, 189–193.
Mullen, G. J., Jelley, R. M., and McAleese, D. M. (1974). A method for measuring soil compactibility in grazed pastures. *Irish J. Agric. Res.* **14**, 43–48.
Muller, S. (1965). Yield and grazing losses in a pasture productivity experiment in Niedehein. *Z. Acker U. Pfl. Bau.* **121**, 171–183.
Mullins, C. E., and Fraser, A. (1980). Use of a drop-cone penetrometer on undisturbed and remoulded soils at a range of soil water tensions. *J. Soil Sci.* **31**, 25–33.
Nash, D. M., and Halliwell, D. J. (1999). Fertilisers and phosphorus loss from productive grazing systems. *Aust. J. Soil Res.* **37**, 403–429.
Nash, D., and Murdoch, C. (1997). Phosphorus in runoff from a fertile dairy pasture. *Aust. J. Soil Res.* **35**, 419–429.
Neal, C., and Jarvie, H. P. (2005). Agriculture, community, river eutrophication and the water framework directive. *Hydrol. Process.* **19**(9), 1895–1901.
Noble, J. C., and Tongway, D. J. (1986). Herbivores in arid and semi-arid rangelands. *In* "Australian Soils: The Human Impact" (J. S. Russell and R. F. Isbell, Eds.), pp. 243–270. University of Queensland Press, St. Lucia.
O'Connor, K. (1956). The influence of wheel and foot treading on soils under grasslands. *N. Z. Soil Sci. Soc.* **2**, 35–37.
OECD. (1982). "Eutrophication of Waters, Monitoring, Assessment and Control." Organisation for Economic Cooperation and Development, Paris.
Oliver, D. M., Clegg, C. D., Haygarth, P. M., and Heathwaite, A. L. (2005a). Assessing the potential for pathogen transfer from grassland soils to surface waters. *Adv. Agron.* **85**, 125–180.

Oliver, D. M., Heathwaite, A. L., Haygarth, P. M., and Clegg, C. D. (2005b). Transfer of *Escherischia coli* to water from drained and undrained grassland after grazing. *J. Environ. Qual.* **34**, 918–925.
Orr, R. J., Parsons, A. J., Treacher, T. T., and Penning, P. D. (1988). Seasonal patterns of grass production under cutting or continuous stocking managements. *Grass Forage Sci.* **43**, 199–207.
Orr, R. J., Rutter, S. M., Penning, P. D., and Rook, A. J. (2001). Matching grass supply to grazing patterns for dairy cows. *Grass Forage Sci.* **56**, 352–361.
Page, T., Haygarth, P., Beven, K., Joynes, A., Butler, T., Keeler, C., Freer, J., Owens, P. O., and Wood, G. A. (2005). Spatial variability of soil phosphorus in relation to the topographic index and critical source areas: Sampling for assessing risk to water quality. *J. Environ. Qual.* **34**, 2263–2277.
Patto, P. M., Clement, C. R., and Forbes, T. J. (1978). "Grassland Poaching in England and Wales." Grassland Research Institute, Maidenhead.
Peeters, A. (2004). "Wild and Sown Grasses. Profiles of a Temperate Species Selection; Ecology, Biodiversity and Use." Blackwell, London.
Piearce, T. (1984). Earthworm populations in soils disturbed by trampling. *Biol. Conserv.* **29**, 241–252.
Prairie, Y. T., Duarte, C. M., and Kaliff, J. (1989). Unifying nutrient chlorophyll relationships in lakes. *Can. J. Fish. Aquat. Sci.* **46**, 1176–1182.
Preedy, N., McTiernan, K., Matthews, R., Heathwaite, L., and Haygarth, P. (2001). Rapid incidental phosphorus transfers from grassland. *J. Environ. Qual.* **30**, 2105–2112.
Pridmore, R. D., Vant, W. N., and Rutherford, J. C. (1985). Chlorophyll nutrient relationships in North Iland Lakes (New Zealand). *Hydrobiologia* **121**, 171–189.
Pykala, J. (2000). Mitigating human effects on European biodiversity through traditional animal husbandry. *Conserv. Biol.* **14**, 705–712.
Rankin, J. D., and Taylor, R. J. (1969). A study of some disease hazards which could be associated with the system of applying cattle slurry on pastures. *Vet. Rec.* **85**, 587–591.
Rasmussen, M. A., Cray, W. C., Casey, T. A., and Whipp, S. C. (1993). Rumen contents as a reservoir of enterohemorrhagic *Escherichia coli*. *FEMS Microbiol. Lett.* **114**, 79–84.
RCEP (1996). "Nineteenth Report: Sustainable Use of Soil." HMSO, London.
Reynolds, B., Emmett, B. A., Thompson, T. R. E., Loveland, P. J., Jarvis, S. C., Haygarth, P. M., Thomas, H. R., Owen, D. H., Roberts, R., and Marsden, T. (2002). "Critical Appraisal of State and Pressure and Controls on the Sustainable Use of Soil in Wales. Report for the Environment Agency and National Assembly for Wales."
Reynolds, S. G., and Frame, J. (2005). "Grasslands: Developments, Opportunities, Perspectives." Science Publishers, Inc, Plymouth.
Rodgers, P., Soulsby, C., Hunter, C., and Petry, J. (2003). Spatial and temporal bacterial quality of a lowland agricultural stream in north-east Scotland. *Sci. Total Environ.* **314–316**, 289–302.
Rook, A. J., and Tallowin, J. R. B. (2003). Grazing and pasture management for biodiversity benefit. *Anim. Res.* **52**, 181–189.
Rook, A. J., Harvey, A., Parsons, A. J., Orr, R. J., and Rutter, S. M. (2004). Bite dimensions of grazing movements by sheep and cattle grazing homogenous perennial ryegrass swards. *Appl. Anim. Behav. Sci.* **88**, 227–242.
Rutter, S. M., Orr, R. J., Penning, P. D., Yarrow, N. H., and Champion, R. A. (2002). Ingestive behaviour of heifers grazing monocultures of ryegrass or white clover. *Appl. Anim. Behav. Sci.* **76**, 1–9.
Rutter, S. M., Orr, R. J., Yarrow, N. H., and Champion, R. A. (2004). Dietary preference of dairy cows grazing ryegrass and white clover. *J. Dairy Sci.* **87**, 1317–1324.
Scholefield, D., and Hall, D. M. (1985). A method to measure the susceptibility of pasture soils to poaching by cattle. *Soil Use Manage.* **1**, 134–138.

Scholefield, D., and Hall, D. M. (1986). A recording penetrometer to measure the strength of soil in relation to the stresses exerted by a walking cow. *J. Soil Sci.* **37**, 165–176.

Schothorst, C. J. (1963). "The Carrying Capacity of Grassland Soils." Wageningen.

Sears, P. D. (1956). The effect of the grazing animal on pasture. *In* "Proceedings of the 7th International Grasslands Congress," pp. 92–101.

Seip, K. L., Jeppesen, E., Jensen, E. P., and Faafeng, B. (2000). Is trophic state or regional location the strongest determinant for the chlorophyll a/TP relationship in lakes? *Aquat. Sci.* **62**, 195–204.

Sharpley, A. N., and Rekolainen, S. (1997). Phosphorus in agriculture and its environmental implications. *In* "Phosphorus Loss from Soil to Water" (H. Tunney, O. T. Carton, P. C. Brookes, and A. E. Johnston, Eds.), p. 467. CAB International, Cambridge.

Sharpley, A. N., and Smith, S. J. (1990). Phosphorus transport in agricultural runoff: The role of soil erosion. *In* "Soil Erosion on Agricultural Land" (J. Boardman, I. D. L. Foster, and J. A. Dearing, Eds.), p. 687. John Wiley and Sons, Chichester.

Sharpley, A. N., Chapra, S. C., Wedepohl, R., Sims, J. T., Daniel, T. C., and Reddy, K. R. (1994). Managing agricultural phosphorus for protection of surface waters: Issues and options. *J. Environ. Qual.* **23**, 437–451.

Simard, R. R., Beauchemin, S., and Haygarth, P. M. (2000). Potential for preferential pathways of phosphorus transport. *J. Environ. Qual.* **29**, 97–105.

Smith, K. A., and Frost, J. P. (2000). Nitrogen excretion by farm livestock with respect to land spreading requirements and controlling nitrogen losses to ground and surface waters. Part 1: Cattle and sheep. *Biosci. Technol.* **71**, 173–181.

Smith, K. A., and Unwin, R. J. (1983). Fertiliser value of organic manures in the UK. *In* "Proceedings Number 221," pp. 31. The Fertiliser Society, NY.

Smith, V. H. (1998). Cultural eutrophication of inland, estuarine and coastal waters. *In* "Successes, Limitations and Frontiers in Ecosystem Science" (M. L. Pace and P. M. Groffman, Eds.). Springer, New York.

Soane, B. D. (1970). The effects of traffic and implements on soil compaction. *J. Proc. Inst. Agric. Eng.* **25**, 115–126.

Soehne, W. H. (1958). Fundamentals of pressure distribution and soil compaction under tractor tyres. *Agric. Eng.* **39**, 276–281.

Svendsen, L. M., Kronvang, B., Kristensen, P., and Graesbol, P. (1995). Dynamics of phosphorus compounds in a lowland river system—Importance of retention and non-point sources. *Hydrol. Process.* **9**, 119–142.

Tamminga, S. (1992). Nutrient management of dairy cows as a contribution to pollution control. *J. Dairy Sci.* **75**, 345–357.

Tanner, C. B., and Mamaril, C. P. (1959). Pasture soil compaction by animal traffic. *Agron. J.* **51**, 329–331.

Taylor, S. A., and Ashcroft, G. L. (1972). "Physical Edaphology. The Physics of Irrigated and Non-Irrigated Soils." Freeman, W H, San Francisco.

Tisdale, J. M., and Oades, J. M. (1982). Organic matter and water-stable aggregates in soils. *J. Soil Sci.* **33**, 141–163.

Trimble, S. W. (1988). The impact of organisms on overall rates within catchments in temperate regions. *In* "Biogeomorphology" (H. Viles, Ed.), pp. 83–142. Blackwell, Oxford.

Trimble, S. W. (1990). Geomorphic effect of vegetation cover and management: Some tine and space considerations in prediction of erosion and sediment yield. *In* "Vegetation and Erosion" (J. B. Thornes, Ed.), Wiley, London.

Trimble, S. W., and Lund, S. W. (1982). Soil conservation and the reduction of erosion and sedimentation in the Coon Creek basin, Wisconsin. *U.S. Geol. Surv. Prof. Pap.* **1234**, 35.

Trimble, T. W., and Mendel, A. C. (1995). The cow as a geomorphic agent—A critical review. *Geomorphology* **13**, 233–253.

Unwin, R. J., Pain, B. F., and Whinham, W. N. (1986). The effect of rate and time of application of nitrogen in cow slurry on grass cut for silage. *Agric. Wastes* **15**, 253–268.

Verstraeten, G., and Poesen, J. (2000). Estimating trap efficiency of small reservoirs and ponds: Methods and Implications for the assessment of sediment yield. *Prog. Phys. Geogr.* **24**, 219–251.

Vickery, P. J. (1972). Grazing and net primary production of a temperate grassland. *J. Appl. Ecol.* **9**, 307–314.

Vinten, A. J. A., Douglas, J. T., Lewis, D. R., Aitken, M. N., and Fenlon, D. R. (2004). Relative risk of surface water pollution by *E. coli* derived from faeces of grazing animals compared to slurry application. *Soil Use Manage.* **20**, 13–22.

Walling, D. E., Collins, A. L., and McMellin, G. K. (2003). A reconnaissance survey of the source of interstitial fine sediment recovered from salmonid spawning gravels in England and Wales. *Hydrobiologia* **497**, 91–108.

Waters, G. R. (1994). Current governmental policy and existing instruments balance on the countryside. *In* "Grassland Management and Nature Conservation" (R. J. Haggar and S. Peel, Eds.), Vol. Occasional Sympousium No. 28, pp. 3–9. British Grassland Society, Aberystwyth.

Watson, S. B., Brownlee, B., Stachwill, T., and McCauley, E. (1999). The use of solid phase microextraction (SPME) to monitor for major organoleptic compounds produced by chrysophytes in surface waters. *Water Sci. Technol.* **40**, 251–256.

Willatt, S. T., and Pullar, D. M. (1983). Changes in soil physical properties under grazed pastures. *Aust. J. Soil Res.* **22**, 343–348.

Wind, G. P., and Schothorst, C. J. (1964). The influence of soil properties on suitability for grazing and of grazing on soil properties. *In* "Proceedings of Eighth International Congress of Soil Science." Romania, Bucharest.