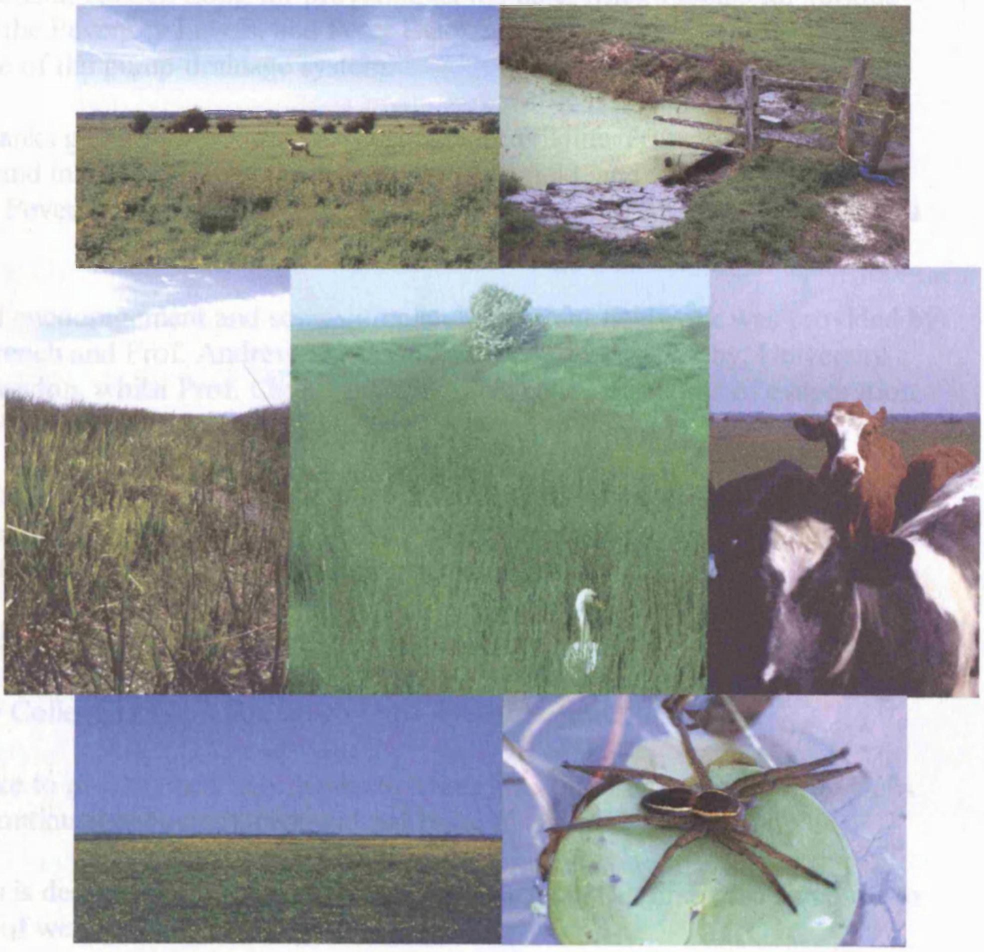
THE HYDROLOGICAL FUNCTIONING OF THE PEVENSEY LEVELS WETLAND



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ABSTRACT

The Pevensey Levels are a wet grassland of national importance in East Sussex, England. The site has been reclaimed from the sea since the Middle Ages, and has traditionally been used for grazing. A purpose-built hydro-ecological model that predicts water levels in ditches and relates them to the hydrological requirements of target species has shown that a traditional water level management regime for grazing is suitable for ditch flora and fauna, the flagship species of nature conservation importance on the site. During this century, the installation of pump-drainage has caused a decline in the range of these species. This causal link has also been confirmed using the hydro-ecological model developed. The Wildlife Enhancement Scheme (WES), a scheme that pays farmers to manage their land in an environmentally sensitive way, has been the main tool employed for the restoration of the site. The WES includes prescriptions to raise water levels. However, a catchment water balance model, quantifying all wetland inflows, outflows and sinks, indicates that the water demand associated with the WES in spring and summer coincides with a period of net water resource deficit. The dimensions of embanked channels, which are employed to feed lowland ditches in the summer are insufficient to provide the storage of winter runoff required to implement the scheme wetland-wide. Micro-meteorological studies also indicate that any attempts to capture winter rainfall in field ditches by raising sluice levels are offset by higher rates of evaporative loss in spring and summer. Consequently, higher water levels in winter do not necessarily lead to higher water levels in summer. In many summers therefore, higher water level targets cannot be attained. This suggests that areas targeted for restoration on the Pevensey Levels should be prioritised to account for water scarcity. Hydrological and ecological monitoring should be undertaken to identify the key areas to be targeted for restoration.

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CHAPTER 1

LOWLAND WET GRASSLAND IN THE UK: HISTORY, HYDROLOGY AND RESTORATION

1.1. Introduction

Lowland wet grasslands are a habitat of considerable ecological importance in the United Kingdom and Europe, the flora and fauna of which is as much a factor of historical anthropogenic intervention, as of their current land-use management. Wet grasslands are therefore a 'cultural' landscape: without agriculture their biological diversity and scenic qualities would not exist (Spellerberg *et al.*, 1991). The term wet grassland incorporates a variety of wetland habitat types, including coastal grazing marshes, floodplain washlands, water meadows and river valley pastures (RSPB *et al.*, 1997). A feature shared by all wet grasslands is their traditional use for the provision of summer grazing and hay for livestock, although many have been increasingly turned to arable production during the latter half of the 20th century (Cook and Moorby, 1993). As a landscape, they are analogous of the Dutch polderlands, areas where the water table was traditionally too high for the intended land use (Smedema and Rycroft, 1983). In order to maximise agricultural productivity, in most wet grassland areas, the relic drainage system has been modified to allow the artificial control of the open water level in response to climatic conditions. Wet grasslands are therefore characterised by the presence of intricate networks of drainage ditches, interlinked by hydraulic structures allowing wetland hydrological functioning to be controlled.

Wet grasslands are prone to flooding in winter, and are therefore closely associated with hydrological systems where periodic or seasonal flooding is a regular feature. As a result, they can be considered 'wetlands' based on the operational definition of these semi-aquatic habitat types provided by the Ramsar Convention (Ramsar, Iran, 1971):

'Areas of marsh, fen, peatland or water, whether natural or artificial, permanent or temporary, with water that is static or flowing, fresh, brackish or salty, including areas of marine water, the depth of which at low tide does not exceed six metres'.

Based on the distinctions provided by the Ramsar Bureau (Davis, 1994), wet grasslands are most appropriately classified as Inland Type 9 wetlands (seasonal freshwater ponds and marshes on inorganic soil, including sloughs, potholes, seasonally flooded meadows and sedge marshes), or Inland Type 12 wetlands (Peatlands: shrub or open bogs and fens) where a peat-based substrate is present.

In recent times considerable concern has been raised over the conservation of wet grassland landscapes at both the national and European levels. Figures of wet grassland loss in the UK are typical of the loss of wetlands in general (Hollis, 1995). Lying in the transition area between truly aquatic and terrestrial habitats, wetlands are frequently affected by human alterations (Hollis and Thompson, 1998). Historically, drainage for agriculture, and other anthropogenic interventions with the hydrological regime, have altered the 'natural' hydrological functioning and led to a decline in both the extent and ecological quality of wet grassland habitats (Joyce and Wade, 1998). During the 20th Century, drainage for agriculture has led to many of the hydrological processes responsible for natural wetland creation being engineered out of existence, a process that has been strongly influenced by technological change. The protection and restoration of wetland sites has therefore focused strongly on the restoration of traditional hydrological management approaches (Wheeler *et al.*, 1995). An understanding of wetland hydrological functioning is fundamental to provide an understanding of the processes that have led to the degradation of individual wetland sites, and to enable their successful restoration.

However, relatively little has been published specifically about the hydrological functioning of wet grassland wetlands (Cook and Moorby, 1993, Denny, 1993, Gilman, 1994). This feature is shared by wetlands in general, where hydrology is generally poorly understood (Lloyd *et al.*, 1993). This is mainly because until relatively recently, wetlands were regarded as wastelands, and were not monitored to the same extent as other types of catchments (Hollis and Thompson, 1998, Al-Khudhairy *et al.*, 1999). To date, hydrological monitoring of wetlands has tended to focus on 'flagship' sites protected by legislation (Cook and Moorby, 1993, Hollis and Thompson, 1998), the designation of which has been traditionally determined by habitat value for bird species. As a result, in many areas where suitable data are available, these have not been compiled on a catchment-wide basis, which is the standard unit of hydrological assessment (Shaw, 1993).

According to Beran (1982), obstacles to the application of a catchment based approach in lowland areas in the UK where data are available include;

- the variety of recording media employed (ledgers, charts, notebooks),
- the variety of data types recorded (water level, pump hours, rainfall),
- the variety of measurement intervals used (irregular entries, daily entries, continuous recorders), and
- the tendency to throw away data by some authorities.

A secondary cause is that hydrological data have been traditionally collected for flood defence purposes. The remit of the flood defence engineer is not related to the historical study of hydrology, but with the response to hydrological conditions in real time. There is therefore a focus on extreme events, and information gathered for flood defence purposes is recorded and deleted soon after (Peter Blackmore, Environment Agency, Pers. Comm).

The main objective of this thesis is to provide an example of the way in which hydrological assessment can be employed to address hydrological management issues in wetland areas. The study focuses on the Pevensey Levels, a wet grassland of national and European significance in southern England. This first chapter provides an overview of the hydrology of wet grasslands in the UK. In doing so, it describes their history, land use and the legislation that currently dictates management in these areas, especially with regards to hydrological management. The chapter also includes a discussion of the biodiversity value of these wetland types, and an examination of the influence of hydrology on habitat ecology and biodiversity. Excellent reviews of many of these issues can be found in Joyce and Wade (1998) and RSPB *et al.*, (1997). This Chapter however, provides a stronger hydrological focus than previous studies, identifying the hydrological processes and issues that characterise wet grassland wetlands to provide a template against which the hydrology of the Pevensey Levels, a lowland wet grassland in East Sussex (UK), can be considered in later Chapters.

1.2. Wet Grassland in the United Kingdom

1.2.1. CHANGES IN EXTENT AT THE NATIONAL LEVEL

The term 'wet grassland' has only recently been widely applied in the scientific literature. This factor, coupled with the broad continuum of wetland types it represents, has complicated the evaluation of changes in the extent of this wetland type. Indeed, losses of wet grassland have only been documented since the war, and particularly over the last 20 years (RSPB, 1996). However, surveys to date have identified wet grassland habitats as one of the most rapidly diminishing in the UK, with direct losses since the 1930s being over 40% by area (RSPB, 1994). 2000 years ago there were probably 2mn ha of lowland floodplain and river delta wetland in England and Wales, an area that has declined to 280,000ha (Newbold, 1998). A survey by the Nature Conservancy Council (NCC) in 1977 concluded that over the previous 35 years the UK had lost 95% of lowland herb-rich grassland. 50% of lowland fells and marshes (Whitby, 1994). 60% of lowland raised mires (Denny, 1993) have also been lost. Losses of neutral grassland, one of the characteristic floral assemblages associated with wet grassland areas (see Section 1.4), have been especially large. Jefferson and Robertson (1994) indicate that only 3 % of the original stock has been left undamaged, with annual rates of loss in the order of 2 - 10 %.

The variety of terms employed above illustrate the present difficulties associated with providing an inclusive definition of wet grassland. Given their mainly agricultural land use, a potential method is the use of agricultural returns to DEFRA (Department for the Environment and Rural Affairs, formerly the Ministry of Agriculture, Fisheries and Food, MAFF). The term 'rough grazing' probably approximates the notion of 'wet grassland' very closely. Land use records therefore possess considerable potential for the provision of baseline data describing wet grassland extent in the UK. For example, Fuller (1987) has reported 'the striking decline in the extent of rough pasture, from 7.2 million hectares in 1932 to 0.6 million hectares in 1984, a loss of 92% that has occurred mainly during the post war period'. These estimates however include chalk grassland and rough upland pastures, although it is possible to discern between upland and lowland components in peat-dominated areas. Peatlands account for 8% of the total land area in the UK, although much of it is in upland areas (Burt, 1995). Indeed, of the 361,690 hectares of peat grassland resource, only 30,000 ha is lowland grassland (Croft and Jefferson, 1994).

The most commonly quoted estimate of wet grassland extent in the UK is that provided by Dargie (1993). Dargie (1993) has identified 219,410ha (2194 km^2) of wet grassland in the UK, although this does not include parcels of land less than 10ha in extent. The survey, commissioned by English Nature (EN), incorporated a wide variety of techniques, including the use of SSSI (Site of Special Scientific Interest) habitat maps, regional Phase I habitat surveys (NCC, 1990), satellite imagery and aerial photography, as well as the identification of the intricate ditch networks that characterise wet grassland areas on 1:25,000 Ordnance Survey (OS) maps. The identification of the artificial drainage networks that are required for hydrological management at all scales are a highly suitable means to both map and delineate wet grassland areas in the UK. In wet grasslands, traditional approaches to wetland classification based on an examination of water inflows, outflows and sinks (Gilvear and McInnes, 1994) are difficult to apply because the hydrology of most of these habitats has been altered to satisfy land use objectives.

There is a close correspondence between the main wet grassland complexes in the UK identified by RSPB *et al.* (1997) and those reliant on 'complete systems for flood defence and land drainage', as defined by Newbold *et al.* (1989) (Figure 1.1). The areas requiring artificial drainage in England and Wales are 15,000 km^2 in extent, and are characterised by the presence of intricate networks of drainage ditches (Newbold *et al.*, 1989). 9000 km^2 require the use of artificial pumping to support local land use (Beran, 1982). That this figure does not compare favourably with the estimate provided by Dargie (1993) is witness to the extent of grassland re-seeding and arabilisation that the existence of such drainage systems has afforded. In many cases, due to the favourable conditions for agriculture artificial drainage measures have afforded, many traditional wet grassland areas can no longer be classified as such, and in official statistics are classified as agricultural land. Data provided by Newbold *et al.*(1989) therefore probably represent the historical wet grassland stock in England and Wales, and the potential future extent of the habitat if all such areas were to be restored.

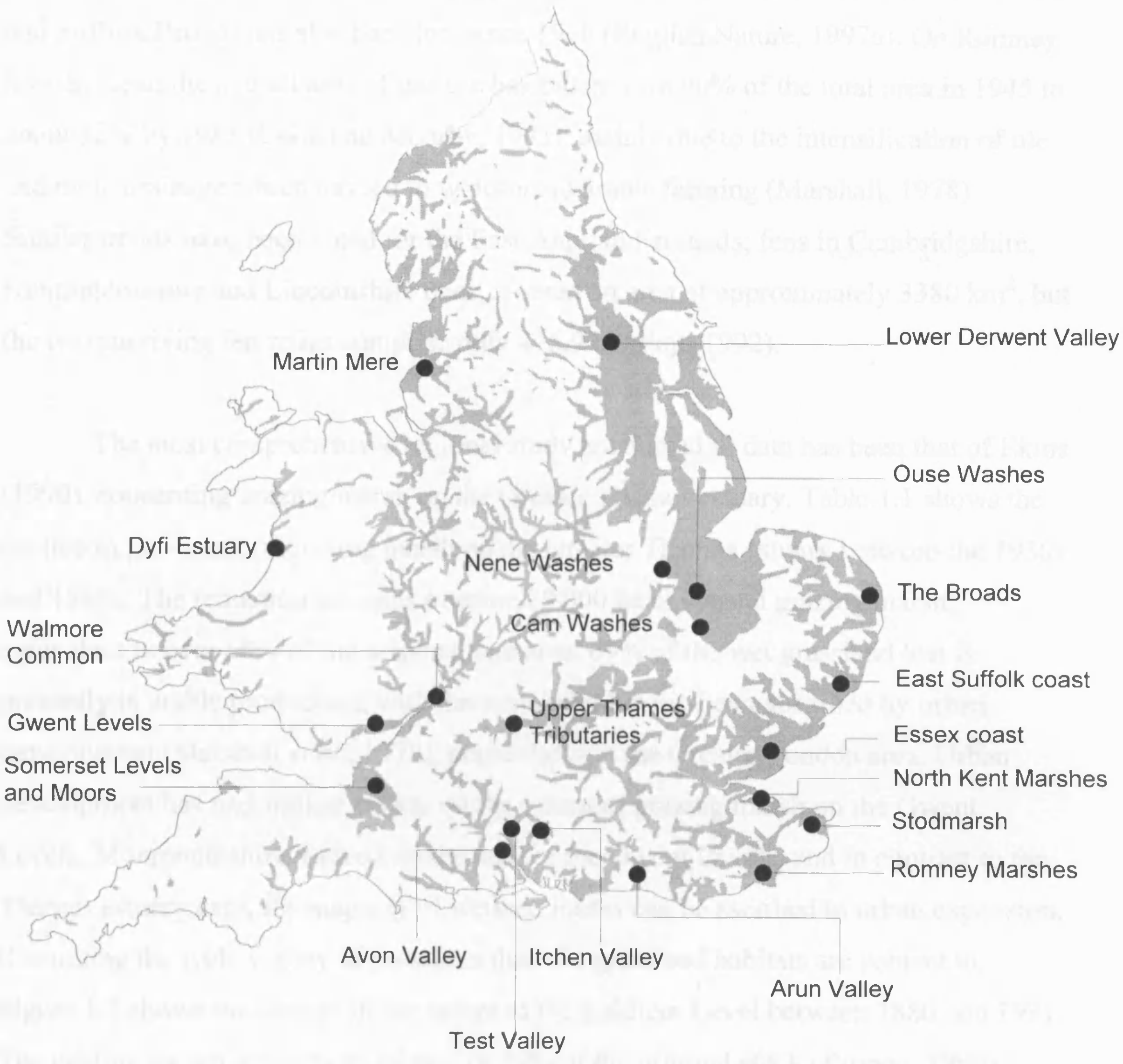


Figure 1.1. The location of major wet grassland complexes in the UK (from RSPB *et al.*, 1997) relative to the distribution of areas reliant on '*complete systems for flood defence and land drainage*', shown as the shaded area (from Newbold *et al.*, 1989).

1.2.2. REGIONAL CHANGES IN EXTENT

Given the difficulties posed by quantifying the decline in extent of the UK wet grassland resource at the national level, most available estimates address declines at the regional or site-specific scales. For example, southwest England is thought to have lost 92% of its wet pastures since 1900 (Denny, 1993). 37% of wet grassland in the Norfolk and Suffolk Broads has also been lost since 1930 (English Nature, 1997a). On Romney Marsh, Kent, the overall area of pasture has fallen from 90% of the total area in 1945 to about 32% by 1985 (Cook and Moorby, 1993), mainly due to the intensification of tile and mole drainage which has led to widespread arable farming (Marshall, 1978). Similar trends have been noted for the East Anglian Fenlands; fens in Cambridgeshire, Huntingdonshire and Lincolnshire once covered an area of approximately 3380 km^2, but the two surviving fen relics comprise only 476.9 ha (Fojt, 1992).

The most comprehensive regional study conducted to date has been that of Ekins (1990), concerning grazing marsh on the Greater Thames estuary. Table 1.1 shows the decline in the extent of grazing marsh on the Greater Thames estuary between the 1930s and 1980s. The reduction amounts to some 28,000 ha of coastal grazing marsh, equivalent to over 60% of the original land area. 69% of the wet grassland lost is presently in arable production, with the remainder having been absorbed by urban development (Marshall *et al.*, 1978), particularly in the Greater London area. Urban development has had similar effects on the extent of grazing marsh on the Gwent Levels, Monmouthshire. Indeed, in the case of the Gwent Levels, and in contrast to the Thames estuary case, the majority of wetland losses can be ascribed to urban expansion, illustrating the wide variety of pressures that wet grassland habitats are subject to. Figure 1.2 shows the change in the extent of the Caldicot Level between 1880 and 1991. The decline shown amounts to 44 km^2 or 39% of the original stock (Rippon, 1996). Landscape loss was initiated by the construction of Cardiff and Newport docks in the 19^{th} Century, covering large areas around the rivers Usk, Rhymney, Taff and Ely. In more recent times, the expansion of Llanwern steelworks and the need to dispose of the large amounts of ash created by these works has resulted in the encroachment of industrial development on a large portion of the site (Rippon, 1996).

	Essex	Greater London	Kent	TOTAL
1930s	25,402	2,767	15,493	43,662
1960s	12,381	1,538	12,898	26,817
1970s	10,542	1,199	9,256	20,997
1980s	7,030	686	7,902	15,618
Loss 1930 - 80 (ha)	-18,372	-2,081	-7,591	-28,044
Change (% of 1930)	-72.3	-75.2	-49.0	-64.2

Table 1.1. Changes in the extent of grazing marsh in the Greater Thames Estuary between the 1930s and 1980s (from Ekins, 1990).

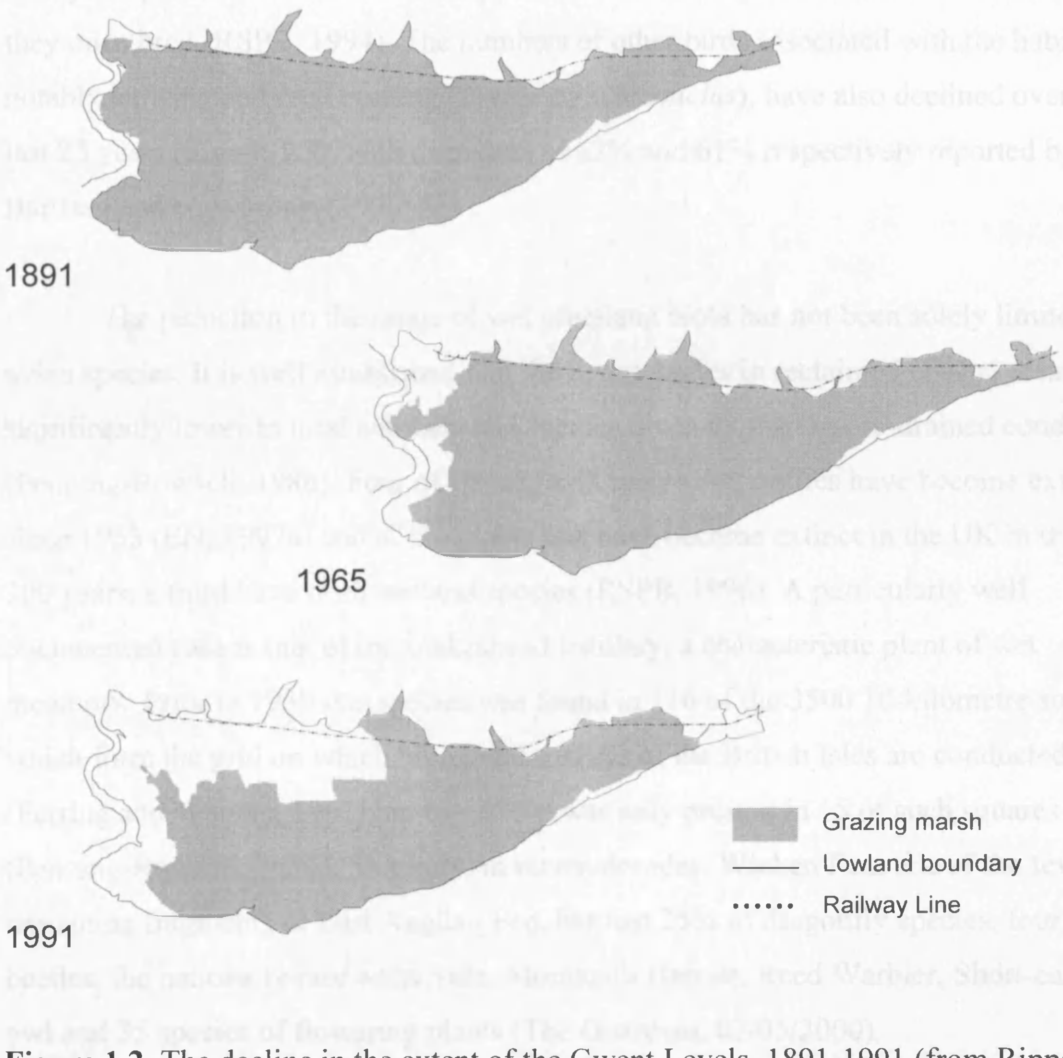


Figure 1.2. The decline in the extent of the Gwent Levels, 1891-1991 (from Rippon, 1996).

1.2.3. BIOLOGICAL EVIDENCE FOR DECLINE

A particularly useful indicator of the decline in the extent and quality of wet grasslands in the UK is to consider the faunal and floral species typical of these habitats. In particular, the unusual cultural importance attached to birdlife in the UK, coupled with the existence of numerous national non-governmental organizations (NGOs) devoted to avian conservation has allowed the loss of bird species to be systematically documented by nature conservation bodies since the late 1960s. The national degradation of wet grassland sites is generally illustrated by overall national numbers of snipe (*Gallinago gallinago*) and redshank (*Tringa totanus*), two characteristic species of traditionally managed lowland grazing marsh (RSPB *et al.*, 1997). Other bird species limited to wet grassland during the breeding season include garganey (*Anas querquedula*), ruff (*Philomachus pugnax*), and black-tailed godwit (*Limosa limosa*) (Joyce and Wade, 1998). Snipe and redshank have disappeared from 60% and 40% of the areas in which they once bred (RSPB, 1994). The numbers of other birds associated with the habitat, notably lapwing and reed bunting (*Emberiza schoeniclus*), have also declined over the last 25 years (Figure 1.3), with decreases of 62% and 61% respectively reported by Bartram and co-workers (1996b).

The reduction in the range of wet grassland biota has not been solely limited to avian species. It is well established that the invertebrates in reclaimed coastal areas are significantly lower in total numbers and species diversity than in pre-drained conditions (Penning-Rowsell, 1986). Four of Britain's 43 native dragonflies have become extinct since 1953 (EN, 1997a) and of the plants that have become extinct in the UK in the last 200 years, a third have been wetland species (RSPB, 1996). A particularly well documented case is that of the snakeshead fritillary, a characteristic plant of wet meadows. Prior to 1930 this species was found in 116 of the 3500 10-kilometre squares which form the grid on which biological surveys of the British Isles are conducted (Perring and Walters, 1962) but by 1970 it was only present in 15 of such squares (Penning-Rowsell, 1986). Similarly, in recent decades, Wicken Fen, one of the few remaining fragments of East Anglian Fen, has lost 25% of dragonfly species, four beetles, the nationally rare water vole, Montagu's Harrier, Reed Warbler, Short-eared owl and 35 species of flowering plants (The Guardian, 02/05/2000).

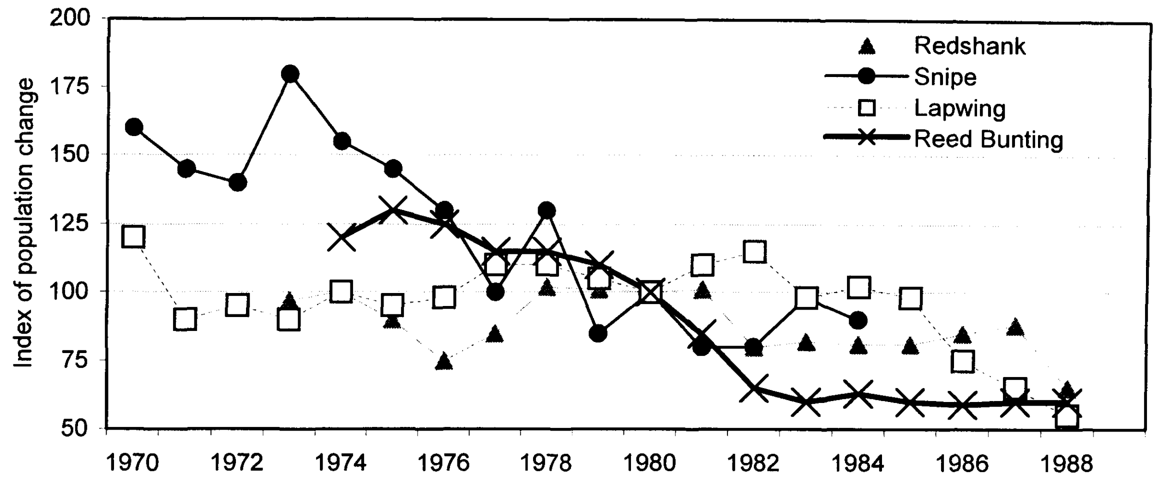


Figure 1.3. Index of population change for characteristic bird species of UK wet grassland between 1962 to 1990 (from RSPB, 1996).

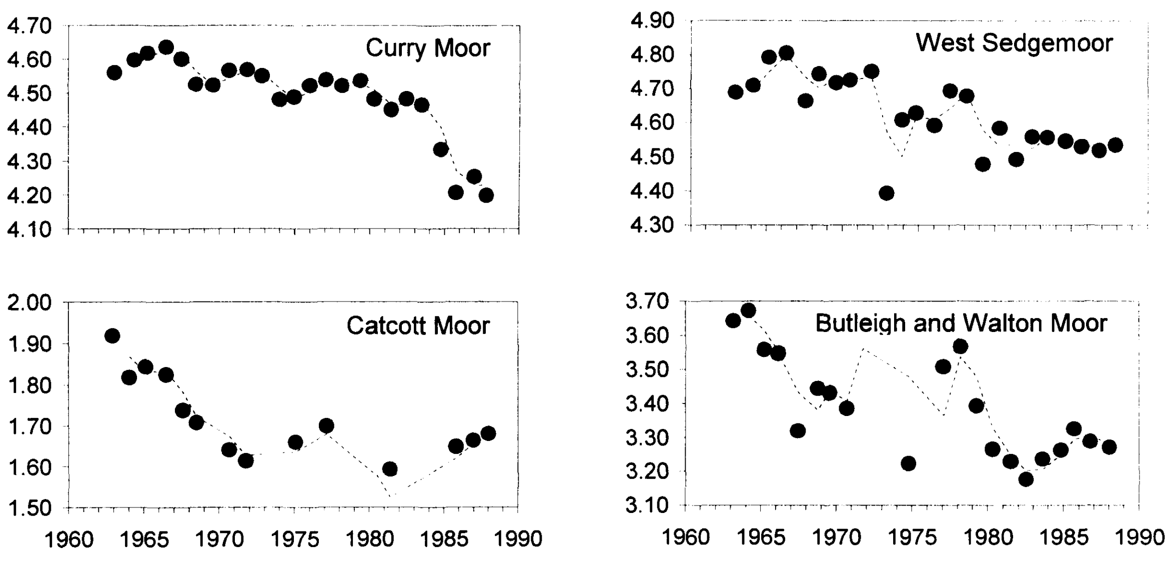


Figure 1.4. Changes in mean annual pump start levels (in m OD) between 1964 and 1987 for four pump-drainage schemes on the Somerset Levels (from Green and Robbins, 1992).

Reductions in the characteristic biota of wet grassland can be ascribed mainly to anthropogenic interference with 'natural' hydrological functioning. Snipe for example, cannot generally probe the ground where the water table is more than 0.2 metres below the surface (Green and Robbins, 1992). In some wet grasslands, drainage for agriculture has lowered groundwater levels by as much as 2 m over the past 25 years (English Nature, 1997). Previous research has linked hydrological management for agriculture to the ecological decline in wet grasslands. For pump-drained areas on the Somerset Moors, Green and Robbins (1992) have found a close association between bird numbers and pump start levels (the water level at which pumping stations become active). Figure 1.4 reproduces pump start levels for four areas on the Somerset Moors between 1964 and 1987. The arabilisation that drainage has afforded, has had a particular influence on floral and invertebrate species inhabiting the ditch system. For example, surveys before and after arabilisation on the Broadland by Driscoll (1983a) have noted considerable declines in species diversity with similar changes recorded in Romney Marsh by Sheail and Mountford (1983). In under-drained areas, a further detrimental feature has been the infilling of ditches whose function has been made redundant (Glading, 1986).

1.3. History of Wet Grassland

1.3.1. PRE-1900

The widespread losses in the extent and quality of wet grassland can be ascribed mainly to processes of agricultural intensification. Human habitation and exploitation of wetlands dates from prehistory (Coles, 1994). In Somerset, hunting platforms were constructed upon the mire surface as early as the Neolithic (Stoneman and Brooks, 1997). Iron age communities made extensive use of marsh landscapes, which were rich in fish and wildfowl, and provided opportunities for salt production and seasonal grazing (Cook and Williamson, 1999). The Roman period saw the initiation of an economy based on the intensive utilisation of these habitats for agricultural purposes (Cook and Moorby, 1993) and coincided with the first attempts to drain and improve soil water conditions for crop production (Thomasson, 1975). Patterns of economic exploitation during the Roman period are found in numerous wet grassland sites in the UK, including the Somerset Levels, the Lincolnshire Fens (Coles, 1994) and the North Kent Marshes (Thompson and Hollis, 1996).

Britain is not known as a land of polders and coastal reclamation although in fact extensive tracts of land are of this character (Beran, 1982b). Rackham (1986) has identified three main stages of wetland reclamation in Britain; Roman, Anglo-Saxon and the 16th and 17th Centuries, when drainage progressed with the expansion of arable farming. It can be argued that the 20th Century post-war period has been a fourth major stage, during which previously reclaimed land has been intensively drained and turned increasingly to intensive, mechanised agricultural production (Section 1.3.2). During Anglo-Saxon times, settlements alongside the fens and marshes of East Anglia were some of the most prosperous in Britain, profiting from the unlimited supplies of fish and edible plants provided by the marshes (Newbold *et al.*, 1989). The use of marshes for salt production was common during Roman times, but became an increasingly important economic activity in areas such as the East Anglian Fens and the Somerset Levels during the Anglo-Saxon period (Cook, 1994). On the Somerset Levels, the earliest known peat cuttings correspond to this period, when peat was used for fuel in the salt production process (Stoneman and Brooks, 1997).

The earliest major drainage schemes were carried out by large landowners who were often ecclesiastical bodies (Mann and Green, 1978). During the Middle Ages monastic institutions played a major role in the reclamation of wetlands and in the Somerset Levels, the Glastonbury and Mulcheney abbeys cut water ways and constructed drains (Cook and Moorby, 1993). Christchurch Abbey was especially active in the reclamation of Romney Marsh (Godwin, 1978) which during the early medieval period (*ca* 1050 to 1250), was used extensively as arable land (Cook and Williamson, 1999). The post medieval period saw the introduction of a Land Drainage Act (1585), which resulted in an acceleration of coastal reclamation works and drainage (Beran, 1987) and witnessed the impact of the great Dutch drainage engineers in areas such as the English Fens (Stoneman and Brooks, 1997). Middle Level, Cambridgeshire, which is still Britain's largest drainage district (130,000 ha) was reclaimed by the Dutch engineer Vermuyden at this time (ICID, 1998).

Throughout the 17th Century wetland drainage had an important social dimension. During the English Civil War in particular, the drainage of the East Anglian Fens proved a contentious issue. The catchment-scale drainage schemes in the East Anglian Fenland caused considerable unrest, culminating in a rebellion in 1653 which resulted in the destruction of major drainage works at Swaffham Bulbeck in 1653

(Hughes, 1991). Indeed, Oliver Cromwell himself was born on the Fens, and lived for a long period of time in the cathedral city of Ely, where he defended the rights of the commoners against aristocratic fen drainers in the late 1630s (Morrill, 1993). Two of Cromwell's deputies and closest friends, Lillburne and Wildman, have been described by Morrill (1993) as *'not averse to breaking the heads or burning the houses of hapless foreign settlers on drained fen'.*

From the 17th Century, drainage frequently led directly to arable land use (Cook and Moorby, 1993). This was achieved by the rapid changes in land drainage technology. Windmills had been used in England since the 12th Century and continued to be erected into the 20th Century (Cook and Williamson, 1999), but in the 17th Century this technique was applied wholesale to drive scoop-wheel pumps where gravity drainage was insufficient to remove excess water (Cook, 1994). By 1600 some 22,000 acres of the Somerset Moors had been drained in this way (Newbold *et al.*, 1989). An alternative to drainage was the storage of winter floodwaters in 'washes' (*e.g.* the Ouse, Nene and Cam Washes). This approach had the advantage of allowing the 'feeding' of crops during times of water scarcity.

From the 17th century onwards, farmers also experimented with various forms of under-drainage (Cook and Moorby, 1993). Mole drainage using a 4 or 8 horse plough was practiced from the 18th Century on clay lands (Thomasson, 1975). It was however the mass production of clay ware cylindrical pipes that lead to installation of field under-drainage in many parts of the UK, and by the late 1840s under-drainage had emerged as the outstanding agricultural improvement of the day (Cook and Williamson, 1999). Further improvement of drainage capabilities were achieved at the field scale by digging existing ditches deeper and creating ridges and furrows (Cook and Moorby, 1993). On heavy clay lands furrows were approximately three metres wide, were aligned down the maximum gradient making them highly topography dependant, and frequently drained into ditches (Cook, 1994). Drainage was supported by loans provided by central government, absorbing at least £27.5 million between 1845 and 1899. This allowed farming to move from being an extractive to manufacturing industry, the essence of the second agricultural revolution (Phillips, 1989).

The invention of the steam powered centrifugal pump, coupled with improvements in the techniques for the installation of under-drainage, produced an incredible increase in arterial drainage in the 19^{th} Century, turning vast areas of summer grazing land into highly fertile arable land (Cook and Moorby, 1993). Throughout the 19^{th} Century technological change made it possible to drain large tracts of marshland for agricultural objectives. In a paper to the Transactions of the Society of Arts, Joseph Glynn (1836) stated that

'few persons are aware of how small a quantity of mechanical power is sufficient to drain a large tract of fenland'.

He was referring to the use of steam power, and its use for drainage by pumping. One of the first areas where this form of drainage was applied was Deeping Fen near Spalding, Lincolnshire, where two steam engines replaced 44 windmills for lifting the water (Glynn, 1836). Witness to the effectiveness of this drainage techniques is the fact that 30 such pumps operated in the Norfolk Broads until at least the 1930s, when they were replaced by diesel units (Newbold *et al.*, 1989).

1.3.2. POST-1900

The dig for victory campaign during, and in the aftermath of, the Second World War caused a particularly large upsurge in agricultural change and drainage. A Land Drainage Act was introduced in 1930. The Agricultural Act of 1937 provided further drainage grants, but also gave financial incentives to farmers to apply lime and slag to grassland. A ploughing subsidy, introduced in 1939, resulted in a large reduction in the area of unimproved grassland in the UK, declining from 5.2 million hectares pre-war, to 3.1 million hectares post-war (Fuller, 1987). The Agricultural Acts of 1947 and 1957 perpetuated these trends by increasing the flow of capital into agriculture, providing price guarantees for all major farm products (Duffey, 1974).

One of the most important capital flows into agriculture at the time was MAFF Capital Grant Aid, initiated in 1942. As part of the Grant Aid scheme any individual or group of landowners could apply for a 50 % loan to cover drainage improvements in their area and additional grants of up to 25 % were available from the European Economic Community (EEC, now European Union [EU]) (Mann and Green, 1978). This included funding to cover the installation of field under-drainage, the deepening and widening of channels and, in some cases the installation of pumping stations. Benefits of under-drainage included yield increases of up to 60 % (Trafford and Massey, 1975), with farmers recouping their investment in a period of between one and six years (Mann and Green, 1978). 1.06 million hectares of land was drained in this way up to 1972 (Thomasson, 1975), so that by 1971 the area of unimproved lowland pastures had declined to 1.8 million ha. In 1978 only 1.00 million ha remained, 18 % of the pre-war total (Fuller, 1987).

Statutory support for drainage and the application of fertiliser resulted in a surge in both drainage and fertiliser application (Figure 1.5). This was coupled with the increasing use of diesel to power pumping stations, allowing the wide-scale drainage of many previously un-reclaimed wet grassland areas. On the Somerset Levels, pumps were installed on the northern end of West Sedgemoor in 1944 to move water out of the moor into the River Parrett (Coles, 1994), lengthening the grazing season by four months (Cook and Williamson, 1999), and in 1930 all existing steam pumps on the Norfolk Broads were replaced by diesel units.

By 1978, 130 x 10^4 ha of land up to 2.5m above flood level had benefited from the 1930 Land Drainage Act, the most extensive regions being the Fens (31 x 10^4 ha), Thorne and Hatfield Moors (3.5 x 10^4 ha) and the Somerset Levels (5.4 x 10^4 ha) (Marshall *et al*, 1978). The drainage effort was greatly aided by increases in the design capacities of pumping units, a factor of the increased pump efficiency that technological change afforded (Beran, 1982b), and the introduction of Internal Drainage Boards (IDBs) to oversee the operation and maintenance of drainage networks. When engines and pumps were first introduced to the Fens of East Anglia, the guiding principle was that they should be capable of removing the equivalent of 6.35 mm of water per day (Beran and Charnley, 1987). After severe floods in March 1947, the standard was raised to 9.5mm and by the late 1950s to 10 mm. In recent times this has increased to 12.5 mm and beyond (Beran, 1982a). Close to 60% of land requiring drainage, as defined by Newbold *et al.* (1989) (Figure 1.1) is drained by pumps. There are over 600 pumping stations in the UK, located mainly on the Lincolnshire and East Anglian fens (368 pumping stations), in Somerset (15 pumping stations), Kent and Sussex (60 pumping stations) (Marshall, 1989).

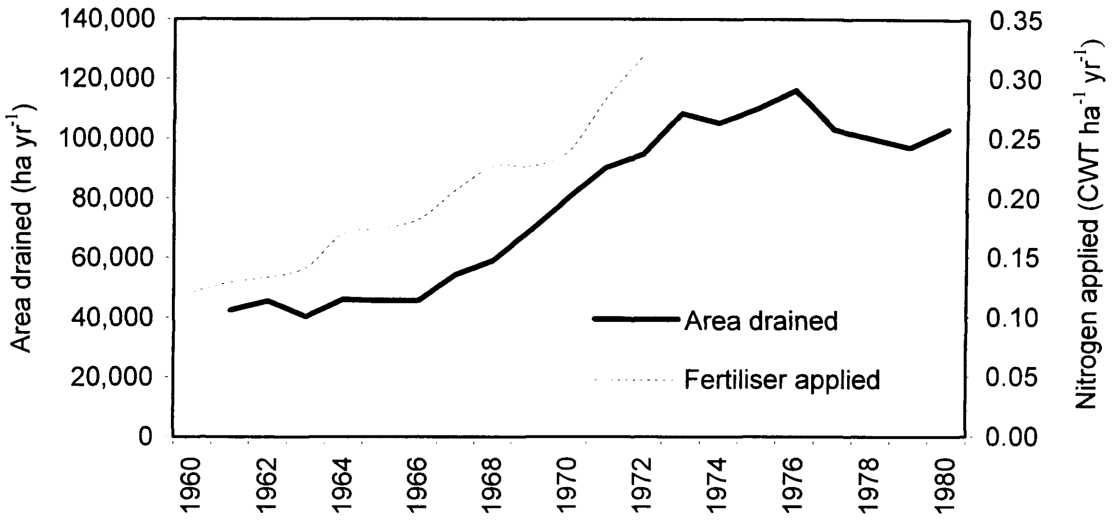


Figure 1.5. Grant aid funded drainage in England and Wales 1960-1980 and the application of nitrogen-based fertilizer during the equivalent period (from Armstrong, 1978 and Duffey, 1974).

Coastal grazing marsh	Fen meadows and *Juncus* pasture	Inland wet neutral grassland
Alopercurus bulbosus	*Erica vagans* (RDB)	*Apium repens* (RDB, GT)
Althaea officinalis	*Gentiana pneumonanthe*	*Carex elongata*
Bepleureum tenuissimum	*Hypericium undulatum*	*Carex tomentosa* (RDB)
Carex divisa	*Lathyrus palustris*	*Fritillaria meleagris*
Cyperus longus	*Lobelia urens*(RDB, GT)	*Oenanthe silaifolia*
Lepidium latifolium	*Peucedanum palustre*	
Trifolium squamosum	*Scorzonera humilis* (RDB, GT)	
	Selinum carvifolia (RDB, GT)	

Table 1.2. Nationally rare and nationally scarce species associated with lowland wet grassland in the UK. GT: globally threatened and declining species, RDB: listed in the British Red Data Book (from Joyce and Wade, 1998).

1.4. Vegetation of wet grassland

Due to a long history of anthropogenic intervention with the natural environment, vegetation assemblages in lowland wet grassland are characteristically semi-natural. From an agricultural perspective, the main aspect that characterises all lowland wet grasslands is their potentially high value in terms of productivity, especially for grazing. For upland areas, Frame (1992) has estimated the potential productivity of grassland swards to be between 27 and 30 tonnes ha^{-1}. However, climatic conditions in lowland areas mean that the season of grass growth is longer and starts earlier, with growth rates up to 100 % greater in lowland versus upland grass swards reported by Lane (1992). Approximately 500 species of vascular plant have been recorded on lowland wet grassland (RSPB, 1994), including floral species that are rare or scarce at the national and international levels (Table 1.2). The floral biodiversity evident is a result of the diversity in micro-habitat types evident within wet grassland complexes. Aquatic, semi-aquatic and dryland ecotones are all present within the typical wet grassland landscape, which is dominated by fields intersected by networks of drainage ditches. Vegetation is also a factor of a long history of 'disturbance': communities commonly exhibit features that illustrate a long history of human intervention with the local environment, including grazing and mowing.

Field vegetation in wet grasslands may be dominated by any combination of 10-15 grass species, the most common being Red Fescue (*Festuca Rubra*), Crested dog's tail (*Cynosurus cristatus*), English Rye-Grass (*Lolium perenne*), Yorkshire fog (*Holcus lanatus*), Sweet vernal grass (*Anthoxanthum odoratum*), Smooth Meadow grass (*Poa pratensis*), Meadow foxtail (*Alopercurus pratenisis*), Common Bent (*Agrostis capillaries*), Yellow Oat Grass (*Trisetum favescens*) and meadow barley (*Hordeum secalinum*) (Countryside Commission, 1991a, RSPB *et al.*, 1997). In improved areas, grass swards are generally characterised by the presence of English Rye-Grass, but also Italian ryegrass (*Lolium multiflorum*), cocksfoot (*Dactylis glomerata*), meadow fescue (*Festuca pratensis*), tall fescue (*Festuca arundinacea*), timothy (*Phleum pratense*), clover, (*Trifolium* spp.) (Fuller, 1987). These may be accompanied by other vegetation types, such as sedges (*Carex* spp.) or rushes (*Juncus* spp.). Over 100 plant species may be present in one field in areas of low intensity grazing (RSPB, 1994), although smaller associations are common. Vegetation communities considered as wet grassland are reviewed in Table 1.3.

NVC Plant Communities		European phytosociological alliances	CORINE biotopes	Notes
(a) Neutral grassland (Jefferson and Grice, 1998) or typical wet grasslands (RSPB, ITE and EN, 1997)				
MG4*	*Alopecurus pratensis-Sanguisorba officinalis*	*Cynosurion*	38.2 Lowland hay meadows	Flood meadows, on alluvial soils, species rich and tall, *Lolium P.* always <10% and usually <5%
MG5	*Cynosurus cristatus-Centaurea nigra*	*Cynosurion*		Old grazed hay meadows, on circumneutral brown soils, species-rich and short
MG6	*Lolium perenne-Cynosaurus cristatus*	*Cynosurion*	38.111 *Lolium perenne* pasture	Dairy and fattening pastures, on circumneutral brown soils and deep loams, species poor, *Lolium P.* always >5% and often >20%
MG7	*Lolium perenne (L Perenne-A. pratensis* grassland)	*Lolio-Plantaginion*	38.111 *Lolium perenne* pasture	Improved and sown swards, on circumneutral brown soils, very species poor without crested dogs tail, *Lolium P.* always >5% and often >20%
MG8*	*Cynosaurus cristatus-Caltha palustris*†	*Calthion*	37.214 *Senecio aquaticus* meadows	Water meadows, on gleyed brown earths, often silty and calareous, or locally peaty, *Lolium P.* always <10% and usually <5%
MG9	*Holcus lanatus-Deschampia cespitosa*	*Calthion*	37.213 *Deschampia cespitosa*	Tussock wet meadows, on circumneutral gleyed brown soils, moerately species rich and tall, *Lolium P.* always <10% and usually <5%
MG10	*Holcus lanatus-Juncus effusus*	*Calthion*	37.217 *Juncus effusus* meadows	Ordinary damp meadows, on gleyed brown earths and alluvial soils, species poor, rushy and tall
MG11*	*Festuca rubra-Agrostis stolonifera-Potentilla anserina*	*Elymo-Rumicion*	37.242 *Agrostis stolonifera* and *Festuca arundinacea* swards	Inundation grasslands, on brown earths and alluvial soils, species poor, rushes rare, sward short, *Lolium P.* always <10% and usually <5%
MG12*	*Festuca arundinacea*	*Elymo-Rumicion*	37.242 *Agrostis stolonifera* and *Festuca arundinacea* swards	On clays and silts
MG13*	*Agrostis stolonifera-Alopercurus geniculatus*	*Elymo-Rumicion*	37.242 *Agrostis stolonifera* and *Festuca arundinacea* swards	Inundation grasslands, on circumneutral silts, very species poor, often in depressions

* Communities considered to be agriculturally unimproved and semi-natural in character
Species richness according to no. of species within 4 m^2 (Rich > 20 spp., Moderate 16-20 spp., Poor 11-15 spp., Very poor <11 spp.)

Table 1.3. National Vegetation Classification (NVC) plant communities considered as lowland wet grassland (from Rodwell, 1992).

NVC Plant Communities		European phytosociological alliances	CORINE biotopes	Notes
(b) Fen meadow (Jefferson and Grice, 1998) or mire grasslands (RSPB, ITE and EN, 1997)				
M22*	*Juncus subnodulosus-Cirsium palustre*	*Calthion*	*Juncus subnodulosus* meadows	On neutral to rather alkaline soils (pH 6-8)
M23*	*Juncus effusus/acutiflorus-Galium palustre*	*Juncion actutiflori*	*Juncus acutiflorus* meadows	On moderately acid (pH 4-6) to neutral mineral soils with high humus content
M24*	*Molinia caerulea-Cirsium dissectum*	*Junco conglomerati-Molinion*	Acid purple moor grass meadows (*Junco-molinium)*	On peats and peaty mineral soil, neutral to mildly acidic
M25*	*Molinia caerulea-Potentilla erecta*	*Junco conglomerati-Molinion*	Acid *Molinia caerulea* (*Junco-molinium)*	Acid to neutral (pH 4-6) peats, or peaty mineral soil
(c)Swamp (Jefferson and Grice, 1998) or swamps and sedge beds (RSPB, ITE and EN, 1997)				
S5*	*Glyceria maxima*	*Phragmition*	*Glyceria maxima* beds	Washland, on nutrient rich, crcumneutral or basic alluvium (pH >6)
S6				Tall sedge meadows, mesotrophic to eutrophic circumneutral mineral soils
S7				Tall sedge meadows, moderately eutrophic, circumneutral to basic mineral soils (pH 6.0-6.8)
S22*	*Glyceria fluitans*	*Spargario-Glycerion*	Small reed beds of fast flowing waters	Floating sweet grass hollows, mesotrophic to moderately eutrophic water on mineral substrates (pH 5-7)
S28*	*Phalaris arundinacea*	*Magnacaricion*	*Phalaris arundinacea* beds	

* Communities considered to be agriculturally unimproved and semi-natural in character

Species richness according to no. of species within 4 m^2 (Rich > 20 spp., Moderate 16-20 spp., Poor 11-15 spp., Very poor <11 spp.)

Table 1.3. Continued.

The dominance of a given species or community is dependant on a number of factors, including hydrology, soil nutrient status and grazing intensity. Augering on Romney Marsh, Kent, has recorded some grass roots at below 1.0 m depth (Cook and Williamson, 1999), although in old permanent pasture close to 90 % of the total root mass is in the top 0.05m of the soil (Voisin, 1959). Hydrology is thus an important control on vegetation, in particular the depth to the water table (Newbold and Mountford, 1997), soil water conditions, and the duration of flooding or waterlogged conditions (Denny, 1993). The hydrological requirements of wet grassland vegetation communities identified in Table 1.3 are summarized in Table 1.4. Table 1.5 shows the dry/wetness ranges of some typical and rare grassland and aquatic species of wet grassland areas. Grazing intensity, and other forms of disturbance, can also have a profound effect on vegetation composition (Voisin, 1959) (Figure 1.6). This is because in grassland communities, a negative relationship between species richness and nutrient availability is evident (Oomes *et al.*, 1996). Grazing removes nutrients from the grassland system, and therefore reductions in grazing intensity can lead to a decline in species richness (Smith and Rushton, 1994).

Aquatic habitats in wet grassland are particularly important in terms of national biodiversity. In coastal marshes, water margins and standing water bodies comprise the habitats of between one and two thirds of the rare and scarce species in the UK (Drake, 1999). On a national basis, the length of drainage ditches exceeds that of all the main rivers, and is comparable to other linear habitats such as hedges (Marshall *et al.*, 1978) (Table 1.6). These drainage channels support some 130 of Britain's 170 species of brackish and freshwater vascular plants and a 20m stretch in a good drainage channel can hold more than 15 aquatic plant species (RSPB *et al.*, 1997).

Aquatic species in wet grassland drainage channels can be crudely classified as emergent, floating or submerged (Figure 1.7). The most botanically interesting ditches are those with a variety of the three types. Ditches dominated by floating species can be poor in submerged species because little light penetrates through the floating biomass, and emergent plants tend to invade open water and compete with submerged vegetation (Newbold *et al.*, 1989). Indeed, some floating species are now becoming a serious nuisance, and changes to water quality and the water regime can result in luxuriant growth at the expense of biodiversity.

	Water table	Flooding regime
MG4*	Soils moist to very locally damp. Free draining above, or sometimes waterlogged (and gleyed) at depth	Winter flooding occasionally persisting into the spring
MG5	Soils moist. Where soil particles are finer, drainage may be impeded - with waterlogging in hollows / furrows	Normally none, standing water in winter is normally associated with other types
MG6	Soils moist but free draining, eliminated by long waterlogging and encouraged by under-drainage	No flooding- or only in very exceptional years
MG7	Soils moist but free draining, eliminated by long waterlogging and encouraged by under-drainage	Where flooded regularly in winter, *Lolium P.* is accompanied by meadow species of *Festuca* and *Alopecurus*
MG8*	Soils constantly damp, due to flood regime or seepage and springs	Deliberately flooded in the past for long period in the winter and spring. This tradition is now rare and the community is found where natural floods occur by rivers
MG9	Soils permanently moist to damp, and with consequent poor aeration	Periodically inundated, *eg.* in furrows - not flooded deliberately
MG10	Soils permanently damp due to ground or surface water	Not normally flooded
MG11*	Soils moist to damp, but free draining	Inundated by fresh or brackish water, but also prone to periods of drying out
MG12*	Soils damp, but free draining	Prone to inundation by brackish water, more rarely tidal water or salt spray
MG13*	Soils damp and sometimes waterlogged	Regularly flooded by fresh water - sometimes for long periods
M22*	Soils moist to damp for most of the year, often due to flushes or springs	Often flooded in winter, very variable in duration, resultant in floral variety
M23*	Soils moist to wet throughout the year, where the drainage is impeded	Not usually flooded
M24*	From fairly moist to quite dry (especially in summer) with little fluctuation in water table or throughput	Very seldom flooded
M25*	Moist but well aerated, often on gentle slopes with lateral water movement	Not usually flooded
S5*	Usually in waterlogged sites, water at soil surface for most of the summer	Regular, very prolonged winter flooding
S6	Continuously waterlogged sites (community also in up to 0.2m of water)	Regular, prolonged winter flooding
S7	Continuously waterlogged sites (community also in up to 0.2m of water)	Regular, prolonged winter flooding
S22*	In grassland waterlogged sites (community also in up to 0.2m of water)	Regular, prolonged winter flooding, often through into late spring/ summer

* Communities considered to be agriculturally unimproved and semi-natural in character

The usage of the terms dry, moist, damp and wet follows that defined by the water indicator (F) values of Ellenberg (1988), where dry = 3, moist = 5, damp = 7, wet = 9

Table 1.4. Hydrological requirements of vegetation assemblages identified as wet grassland in the National Vegetation Classification of Rodwell (1982) (from RSPB *et al.*, 1997).

Species	Dry	Preferred	Wet
*Agrostis stolonifera**	-0.10	0.00	+0.05
*Molinia caerulea**	-1.00	-0.50 -0.25	0.00
*Alopercurus pratensis**	-0.80	-0.50	-0.20
*Alopercurus geniculatus**	-0.20	-0.10 0.00	+0.10
*Poa trivialis**	-0.65	0.00	+0.05
Althaea officinalis‡	-0.80	-0.20	+0.30
Carex divisa‡	-0.30	0.00	+0.10
Cyperus longus‡	-0.15	-0.10 0.00	+0.15
Lathyrus palustris‡	-0.90	-0.60 0.00	+0.15
Peucedanum palustre‡	-1.00	-0.50 -0.20	0.00
Potomageton natans†	+0.02	+0.50 +1.00	+1.25
Rananculus aquatilis†	0.00	+0.30 +0.75	+1.50
Carex aquatilis†	-0.40	+0.10	+0.30
Juncus bulbosus†	+0.10	0.00	+0.80
Phragmites australis†	-1.00	-0.20 0.00	+0.50

- indicates water table below ground level, + indicates water level above ground level (depth of water)

Table 1.5. Water table dry/wetness ranges (in m) of some typical*, rare grassland‡, and aquatic† species of wet grassland (from Newbold and Mountford, 1997).

Habitat Type	Length (km)
Main River	30,571
Canal	3,218
Main Drainage Channel	32,180
Subsidiary Drainage Channel	96,540
Hedges	576,000

Table 1.6. Length of linear habitat in England and Wales (from Marshall, 1976).

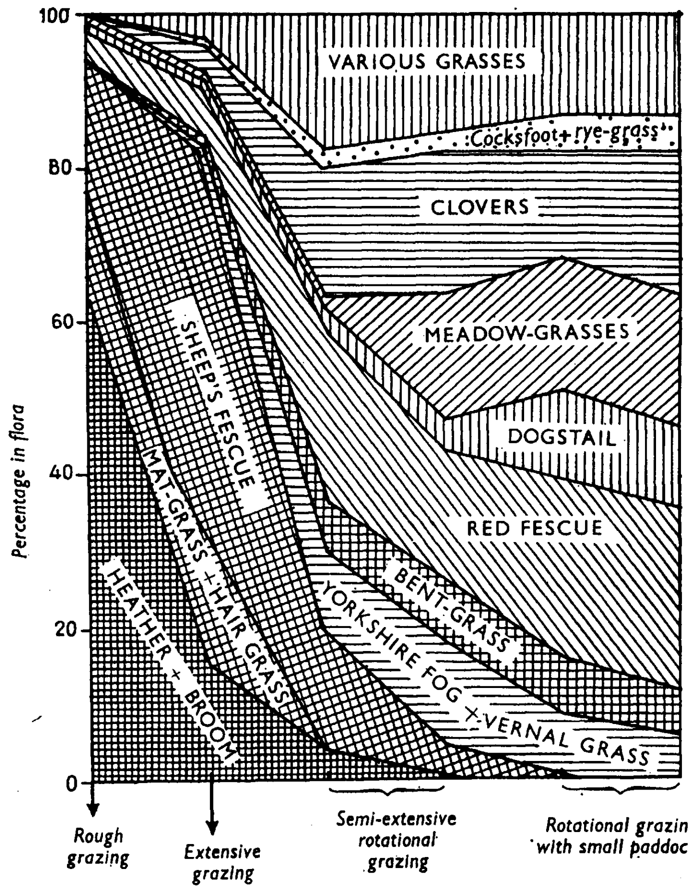


Figure 1.6. Influence of grazing intensity on the composition of a deteriorated grass sward, Rengen, Germany (from Voisin, 1959).

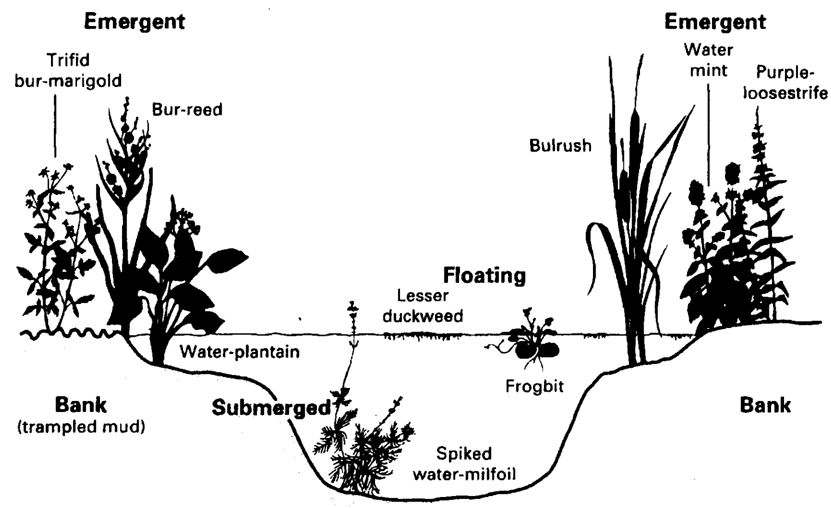
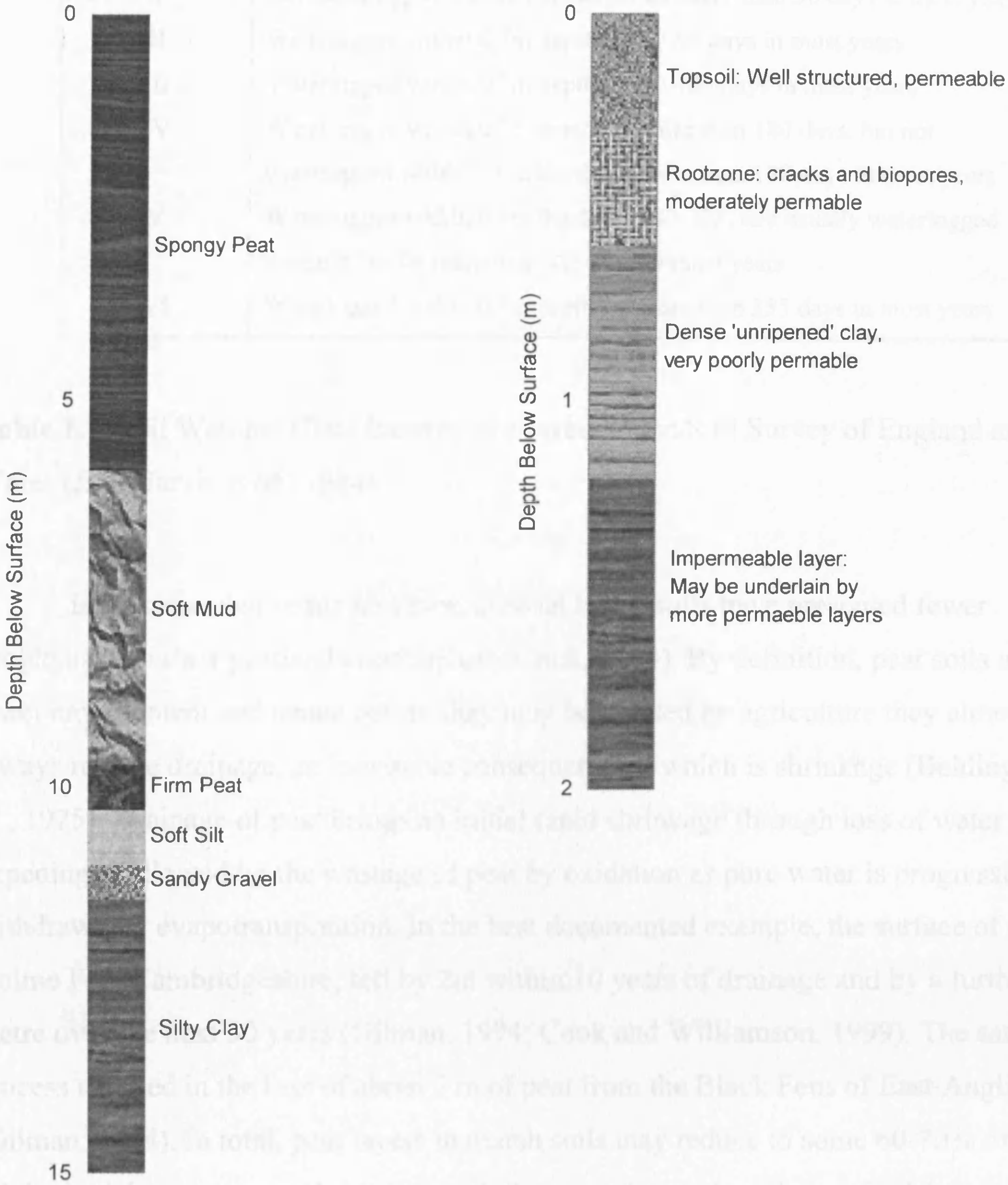


Figure 1.7. Drainage channel profile illustrating the different growth forms and habitats of aquatic and semi-aquatic vegetation (from RSPB *et al.*, 1997).

1.5. Soils of wet grassland

The commonest mineral soils of reclaimed marshes have a textural range comprising sandy clays, silty clays, clays, silty clay loams and silt loams (Cook and Moorby, 1993). Organic soils may be peats, fen peats where calcareous or acid peats where formed in bogs (Cook and Moorby, 1993). Due to historic variations in sea level, mineral and organic soils can be present within the same soil profile (Figure 1.8.a): fen peat accumulates in sheltered freshwater lagoons (Cook and Williamson, 1999), with clays and silts more commonly deposited under marine, or floodplain situations.

Water movement through clay soils can be complex due to the interaction s between large soil cracks and smaller pores (Burt, 1995). In clay soils, the lower horizons are characteristically gleyed, indicating that they are commonly waterlogged for some proportion of the year. The duration and degree of waterlogging has been classified by a system of 'wetness' classes, grading from Wetness Class I (well drained) to Wetness Class VI (almost permanently waterlogged within 0.4 m depth) (Jarvis *et al.*, 1984). Definitions of each of the wetness classes are given in Table 1.7. Soils are intrinsically linked to the hydrological system, determining their chemical, biological and physical conditions (Philipps, 1995). Therefore, any changes to soil hydrology must be carefully considered as they can be detrimental to soil structure and may lead to reductions in agricultural productivity. Soils reclaimed from marine clays can be adversely affected by sodium content: when clay is deposited by salt water, it takes up a card house structure but as this pore water is drawn out the card-house structure collapses, reducing the aeration and water holding capacity of the soil (Cook and Moorby, 1993). A seasonal cycle of drying and wetting is evident in clay soils, causing annual shrinkage and swelling, with changes in land level of up 0.025 m having been reported by Trafford and Massey (1975). This leads to the cracking of the soil which is important for the maintenance of appropriate soil conditions: macropores in clay soils provide a preferential pathway for the flow of water and air into impermeable soil layers.



(a) Somerset Levels (b) Clay soil in a reclaimed area

Figure 1.8. Two examples of stratigraphic sequences in wet grassland areas, reproduced from (a) Gilman *et al.* (1990) and (b) Smedema and Rycroft (1983).

Wetness Class	Duration of waterlogging
I	Not waterlogged within 0.7m depth for more than 30 days in most years
II	Waterlogged within 0.7m depth for 30-90 days in most years
III	Waterlogged within 0.7m depth for 90-180 days in most years
IV	Waterlogged within 0.7m depth for more than 180 days, but not waterlogged within 0.4 m depth for more than 180 days in most years
V	Waterlogged within 0.4m depth for 180-335, and usually waterlogged within 0.7m for more than 335 days in most years
VI	Waterlogged within 0.4m depth for more than 335 days in most years

Table 1.7. Soil Wetness Class Indexes employed by the Soil Survey of England and Wales (from Jarvis *et al.*, 1984).

In management terms however, alluvial based soils have presented fewer problems than their peatland counterparts (Cook, 1994). By definition, peat soils arise in a wet environment and hence before they may be utilised by agriculture they almost always require drainage, an inevitable consequence of which is shrinkage (Belding *et al.*, 1975). Drainage of peat brings an initial rapid shrinkage through loss of water (ripening), followed by the wastage of peat by oxidation as pore water is progressively withdrawn by evapotranspiration. In the best documented example, the surface of Holme Fen, Cambridgeshire, fell by 2m within 10 years of drainage and by a further metre over the next 30 years (Gilman, 1994; Cook and Williamson, 1999). The same process resulted in the loss of about 5 m of peat from the Black Fens of East Anglia (Gilman, 1994). In total, peat layers in marsh soils may reduce to some 60-70% of their original thickness upon reclamation, with the typical covering of organic debris mostly disappearing in a matter of a few years after drainage (Smedema and Rycroft, 1983). The rate of wastage is dependant on the water table levels maintained following reclamation. Mirza and Irwin (1964) have quoted a range of subsidence rates ranging between 0.010 and 0.122 m yr^{-1} associated with shallow and deep water tables respectively, highlighting the importance of adequate hydrological management following drainage for the protection of organic soils from shrinkage and wastage.

In the context of agricultural development, research to characterise soil physical parameters has focused primarily on the determination of hydraulic conductivity (*K*). *K* is an important drainage parameter as it will determine the spacing of under-drainage and ditches installed as part of any drainage network (Smedema and Rycroft, 1983). Because of the variety of soil forming agents evident, values of *K* for wet grassland encountered in the literature are variable. In general though, values of *K* reported for peat soils are higher than those for clay substrates. In the UK, values of *K* of 0.960 md^{-1} for peat soils and 0.024 md^{-1} for alluvial clay soils are quoted by Armstrong (1993) as typical. However, there is usually spatial variability of soil properties (Youngs, Leeds-Harrisson and Chapman, 1989) and natural compaction at depth is common on many clay soils (Beran, 1982a), a feature that is exacerbated by trampling by cattle or the use of heavy machinery when the soil is wet. In the case of peat-based soils, the spatial variability of soil physical properties is especially apparent since *K* is strongly influenced by parent material as well as the degree of humification. For example, the hydraulic conductivity of fibrous peat can be more than an order of magnitude higher than that of amorphous peat (Burt, 1995) and *Phragmites* and *Carex* peats possess considerably higher hydraulic conductivities than *Sphagnum* peats (Belding *et al.*, 1975).

Table 1.8 reviews some of the estimates available for hydraulic conductivity in wet grassland areas. Variations apparent both between and within sites illustrate the importance of field experiments for the determination of *K* for drainage design, as stated by Armstrong and Tring (1980). Field measurement of *K* is also of vital importance in modeling field scale hydrology in wet grassland. Indeed, all current water table models for wet grassland (Armstrong, 1988, 1993, Youngs *et al.*, 1991) identify this parameter as a crucial component for accurate modeling results. This issue, and the hydrological models currently available or the simulation of wet grassland, are considered in detail in later chapters of this thesis.

Source of data	Location	Hydraulic Conductivity (md^{-1})	Notes
Peat-based wet grassland soils			
ADAS (1994)	Norfolk Broads, Norfolk	48.45 – 0.0966 308-0.226	Range of mean *K* values in 6 fields Maximum range in an individual field
Armstrong (1993)	Unspecified	0.96	Typical of peaty soils in the UK
Armstrong and Rose (1998)	Halvergate marshes, Norfolk	100 (surface) 0.1 (> 1m below surface)	Alluvial clay Newchurch series soils that typify reclaimed alluvial soils (Cook and Williamson, 1999)
Bradley and Brown (1989)	Narborough Bog, Leicestershire	0.3 (woody peat) 10 (*Phragmites* peat)	Values employed in hydrological modelling studies
Youngs *et al.* (1989)	Somerset Levels	0.75 – 1.12	Peaty soil
Clay-based wet grassland soils			
Gavin (2001)	North Kent Marshes, Kent	2.77×10^{-5}	Wallasea
Armstrong (1980)	Unspecified	1.268 (mean of 14 sites in England) 0.442 (mean of 21 sites in England)	Marine alluvium (Wentlooge, Waveney series) Riverine alluvium (Compton, Fladbury, Hollington, Kingsland, Pinsley, Stixwold, Wyre series)
Armstrong (1993)	Unspecified	0.96 0.024	Typical of peaty soils in the UK Typical of clay soils in the UK
Bradley and Brown (1989)	Narborough Bog, Leicestershire	0.1 (silty clay)	Values employed in hydrological modelling studies
Childs *et al.*(1957)	Romney Marsh, Kent		*Journal of soil science 8 pp 27*
Cook and Moorby (1993)	Unspecified	1.7×10^{-6} 100	Pre-reclamation marsh clay Post-reclamation marsh clay (due to cracking and ripening)
Giraud *et al.*, 1997	Moeze Marsh, Charente-Maritime, France	0.25 0.18 0.02	Well-drained clay soils Medium drainage clay soils Poorly drained clay soils
Trafford and Massey (1975)	Wicken Fen, Cambridgeshire	0.003	

Table 1.8. Values of hydraulic conductivity (*K*) reported for wet grassland soils (from a variety of sources).

1.6. Hydrology of Wet Grassland

1.6.1. THE DRAINAGE NETWORK: STRUCTURE, FORM AND CHARACTER

Human intervention in wet grasslands has created a range of modified wetland landscapes which share many common characteristics, the most notable of which is the need for hydrological management at all scales (Stoneman and Brooks, 1997). Appropriate conditions for agriculture have been generally provided by constructing channel networks which are superimposed upon the 'natural' hydrological system. The relative sinuosity of the channels is a useful means of differentiating between pre-reclamation and drainage channels. In the Middle Ages, drainage ditches were constructed to follow the natural drainage lines of the primary marsh, contrasting with the rectilinear drains of 17^{th}-19^{th} Century reclamation, representing large scale planning (Cook, 1994). Drainage density in pre-reclamation wetlands tended to be low and a key feature of reclamation efforts was a further increase in the drainage density, reflecting the need to reduce water residence time.

The objective of drainage is to maximize agricultural productivity. This requires a seasonal approach to management. A typical drainage network is shown in Figure 1.9 and will consist of the following components:

- open field, or subsurface drains,
- ditches,
- main ditches,
- embanked channels and
- pumping stations. (Schulz, 1980)

During the winter months, the drainage network is operated to reduce the duration of inundation, reduce anoxia in soil, improve trafficability and reduce poaching by using pumping stations to draw water from smaller field scale ditches into larger embanked channels. During the spring and summer, larger channels can be used to store water. Water can be used later in the season to help irrigate grass or arable crops.

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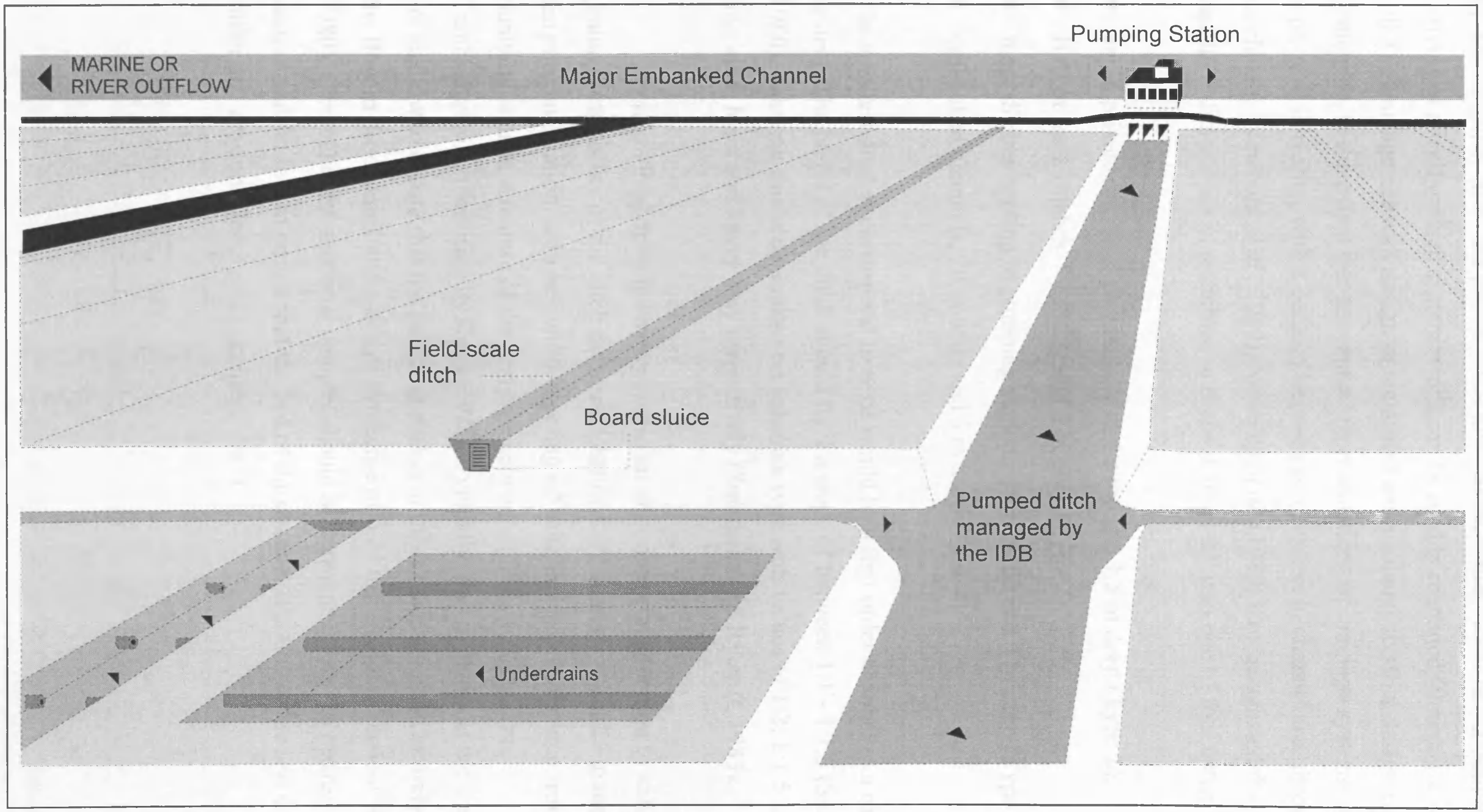

Figure 1.9. Schematic view of the water management system (from Schulz, 1980). Arrows indicate predominant flow direction.

The design principles employed have given rise to hierarchical drainage networks, with smaller ditches combining into higher order channels in the process of centralisation of flow and convergence towards the outlet, constructed with the aim of arriving at a well ordered system which can be readily managed (Smedema and Rycroft, 1983). Drainage network design in reclaimed areas adheres to strict design rules and principles, to the extent that conceptual representations of drainage systems are closely replicated in reality, with a definite stepwise progression in channel size through the catchment (Newbold *et al.*, 1989). Newbold *et al.* (1989) have suggested that ditches and their dimensions in reclaimed or drained areas fall into one of four categories:

- small private ditches 3 metres wide at the top and 1.5 m deep (Type 1),
- IDB or main ditches, 8 m wide and 3 m deep (Type 2),
- main ditches leading to pumping stations, 10 m wide and 3 m deep (Type 3) and
- embanked channels, 20 m wide and 5 m deep (Type 4).

The smallest ditches correspond to what would be a first order channel in a natural system. The banks of the ditch should have a slope of between 1:1 - 1:1.5 (Schulz, 1980), although this is dependant on substrate type with values of 1:2, 1:1.5 and 1:1 for fine sand, loam and heavy clay respectively (Smedema and Rycroft, 1983).

Longitudinal drain geometry is also an important consideration to enhance the drainage capability of the ditch network. Channels are characteristically graded towards the pumping station, with bed width tapering off with increasing distance from the pumping station, a feature of the drainage network on Newborough Fen, Cambridgeshire, identified by Reed (1985). Typically, the gradients of these channels are shallow to ensure that the pumping station or other drainage point entirely controls the flow in the channel and a design discharge of 0.25 ms^{-1} is not exceeded (Schulz, 1980). Channel cross-sectional velocity should be constant across the entire cross-section and set low to ensure that the bed of the channel remains stable and does not suffer any erosion (Beran and Charnley, 1987).

1.6.2. RUNOFF

The average runoff in drained lowland areas is higher than in natural catchments and may be ascribed to the greater ability of a fen catchments to intercept and evacuate rainfall due to the dense stream network (Beran, 1982a; Burt, 1995; Table 1.9). Runoff magnitude is an important component of the design of drainage networks in lowland areas, determining pump capacity and the dimensions of ditches and channels constructed. By allowing the calculation of the likely volumes of water associated with rainfall events of design return periods, the ditch network can be designed to provide the required water storage capacity to limit inundation events (Mann and Green, 1978). However, few data describing runoff coefficients for lowland areas in the UK are available. For the design of drainage system in Dutch polderlands, a runoff coefficient of 80% is employed (Schulz, 1980). In the UK, a general approach to determine runoff coefficients has been the application of the Flood Studies Report (FSR) method (Natural Environment Research Council [NERC], 1975) (see Sutcliffe, 1978). In the FSR method, the percentage runoff, or runoff coefficient (R_c) is given by

$$R_c = SPR + 0.22\,(CWI - 125) + 0.1\,(P - 10) \qquad \text{(Equation 1.1)}$$

where SPR is the standard percentage runoff predicted from the soil type of the catchment, CWI is the catchment wetness index obtained from a five day Antecedent Precipitation Index (API) and the soil moisture deficit and P is storm rainfall.

The FSR method has been employed by Beran and Charnley (1987) on Newborough Fen, Cambridgeshire. Monitoring has indicated that the ditch network responds to flood inflows more or less as one body, with water levels at the pumping station rising almost as soon as at remote sites (Beran, 1982a). This unity of response is thought to be attributable to the shallow gradient of the main drain and the strong influence that the local water table exerts on the catchment response to rainfall (Reed, 1985). For 10 individual rainfall events the values of R_c provided by this study are reproduced in Table 1.10. Results presented support the validity of the approach in providing realistic representations of runoff magnitude in lowland areas. Actual runoff coefficients obtained were associated with a mean value of 32%, in close accordance with the value of 29% derived from the FSR methodology (Beran, 1982a). This approach has also been employed on the Willingdon Level, East Sussex, with R_c values of 42% reported by Binnie and Partners (1988) as appropriate for this alluvial clay area.

The influence of drainage on runoff generation has received considerable attention in the scientific hydrological literature. Authors have concluded that drainage does indeed have an important influence on hydrology (Dunn and Mackay, 1996), although two contradictory mechanisms have been proposed. Beran (1982b) has suggested that by increasing drainage density, engineered drainage systems posses shorter lag-times and increased peak flows relative to their natural counterparts (Beran, 1982b). For example, Giraud *et al.* (1997) report values of discharge 27% greater for under-drained areas in a French coastal marsh relative to un-drained areas at the same location. This contrasts with findings by Iritz *et al.* (1994), that suggest that drainage reduces runoff and peak flows due to reductions in water table levels across the lowland catchment.

This latter mechanism is supported by work conducted on the North Kent Marshes by Al-Khudhairy *et al.*(1999), who, based on a modelling approach, have reported increases in peak flows following the removal of the sub-surface drainage system. Equivalent results have been presented by Johansson (1993) based on studies of Finnish wetland soils, although smaller increases of peak and low flows in the region of 3.5% in drained versus un-drained areas are reported. However, most evidence suggests that drainage increases the volume of discharge but flattening the flood peak (Penning-Rowsell *et al.*, 1986). This gives rise to the characteristic shape of the unit hydrograph in lowland areas, which is generally trapezoidal in form (Beran and Charnley, 1987). Decreases in lag time due to drainage are ascribed by Dunn and Mackay (1996) to increases in the speed of runoff generation, and the proportion of total runoff contributing to sub-surface flows. In artificially-drained, lowland areas the lag time, the interval between the rainstorm centroid and the hydrograph peak is commonly around 24 hours (Beran and Charnley, 1987).

District Name	Area (km^2)	Channel Length (km)	Drainage Density ($km\ km^{-2}$)
Romney Marsh	230	84	0.37
Tilbury Marshes	51	24	0.47
The Fen District	3126	4426	1.42
Lincolnshire coast	190	930	4.89
Beverley and Holderness	41	756	18.44
Over Wyre	50	109	2.18
Fylde Marshes	64	31	0.48
Southport Marshes	143	250	1.75
Monmouthshire Moors	80	199	2.49
South Gloucestershire Levels	13	354	27.23
Walton-Gordano/ Yeo Valleys	90	384	4.30
River Brue/ River Axe Valleys	240	799	3.33
River Parrett Valleys	186	512	2.75

Table 1.9. Drainage density in selected UK coastal marshes (from Marshall *et al.*, 1978).

Event Date	Rainfall (mm)	Predicted R_c (%)	Observed R_c (%)
1st January 1980	18.0	26.8	59.2
31st January 1980	8.0	28.6	33.3
17th March 1980	32.5	30.6	67.0
14th August 1980	42.5	7.9	5.0
19th December 1980	6.5	20.4	33.6
23rd April 1981	78.5	34.0	30.0
15th March 1982	25.5	30.1	28.0
25th June 1982	35.5	21.5	20.0
14th November 1982	27.0	29.3	26.8
18th May 1983	51.5	32.5	35.0

Table 1.10. Runoff coefficients for individual flood events on Newborough Fen, Cambridgeshire (from Beran and Charnley, 1987).

1.6.3. DITCH WATER LEVEL MANAGEMENT

The main purpose of these ditch networks are to convey winter runoff to channels where it may be either discharged, or stored for re-distribution during the drier summer months. Hydrological management in wet grasslands is therefore strongly focused on the control of the storage component of the wetland water balance. Traditionally irrigation was used to flush pastures with nutrients at the start of the growth season, protecting from frosts or warming the soil during early spring (Cook, 1994, RSPB, 1994). The present century however, has seen a large increase in the localised control of ditch water levels in wet grassland areas (Cook and Moorby, 1993), designed to lower the water table to allow more intensive grazing and arable cultivation (Penning-Rowsell et al., 1986). In 'natural' systems, the driving forces behind water level variations in wet grassland areas are seasonal in nature, and the balance between rainfall and evaporation is the most important variable (Thompson and Hollis, 1996). This leads to water level maxima in winter with minima in the mid- to late summer, as in natural riverine systems, although throughout this century water level management for agriculture has become increasingly superimposed upon the natural hydrological regime.

Agricultural objectives have increasingly required that the drainage system be operated as an integrated unit (Garcia *et al.*, 1992). This has been especially the case because reductions in yield due to moisture deficits during the summer have become relatively more important following drainage (Prak, 1988). Cannell *et al.*(1984) report reductions of 7 and 9 % in the yield of winter barley and winter wheat respectively due to drought in a clay soil. Although these reductions in yield are a third of those associated with waterlogging (see Section 1.6.5.), Meteorological Office and DEFRA/MAFF bulletins indicate that crops in areas such as Southern England are under drought stress 8 years out of 10 (Beran and Charnley, 1987), highlighting the economic importance of providing suitable water levels for crop irrigation. For perennial ryegrass (*Lolium Perenne*), differences in daily productivity in the region of 10% in irrigated versus un-irrigated plots are reported by Frame (1992). Where grazing is the main form of land use, an added feature of the management of ditch water levels in summer is that subsidiary channels contain sufficient water to act as 'wet fences' (field boundaries) and provide accessible watering places for livestock.

In contrast, during the winter months water levels should be maintained to provide sufficient storage capacity in the drainage system for potential flood events (Smedema and Rycroft, 1983). This practice ensures the access and workability of the land, by reducing the incidence of waterlogging and especially flooding, which is associated with crop damage occurring over a much shorter time scale than waterlogging (Figure 1.10). As a result of these agricultural objectives, water level management in wet grassland areas for agriculture has effectively reversed the 'natural' seasonal trends in water level, promoting higher levels in the summer than the winter, coupled with an associated reduction in mean ditch water levels on an annual basis (Figure 1.11). In grazed areas in the UK, general practice is to maintain water levels at 0.4 m from the field surface from April to November, and 0.75 m from the field surface at other times of year (Spoor and Gowing, 1995) (Figure 1.11). In some areas, this range can be satisfied, as in the Somerset Levels (Youngs *et al.*, 1989) or Llyn yr Wyth Eldion, Cors Erddreiniog (Gilman, 1994). However, due to the important influence of evapotranspiration in summer, the annual range in water levels is generally higher, with a maximum range of 1.0m (mean 0.53m) reported for the North Kent Marshes based on water level monitoring over an eight year period (Hollis *et al.*, 1993).

The specific water level regime is however ultimately dependant on the value of the crop grown, it's susceptibility to waterlogging and inundation, the size and organisation of the drainage system, and the climatic conditions in individual years. Probably the simplest distinction that can be made in this respect relates to whether 'high' (*eg.* Cereals) or 'low' (*eg.* grass) value crops are grown. Thus, for arable crops in the Netherlands van Bakel (1988) suggests ditch water levels of between 0.9m below field level between April and September and 1.4m for the rest of the year as appropriate (Figure 1.11), with a water depth of at least 0.7m in the ditch (Ritzema, 1994). In many areas, especially where arable agriculture is the main form of land use, target water levels are maintained by pumping stations. A typical pumping station is shown in Figure 1.12. The hydrology of pumped-drained ditches is dominated by large, rapid fluctuations in water level when the pump is operational, although these variations are generally smoothed out within 3km of the pumping station (Beran, 1982a, Marshall, 1989). The specific range of water level variations are a factor of the pump start and stop water levels, determined by electrodes that can be varied according to seasonal water level objectives.

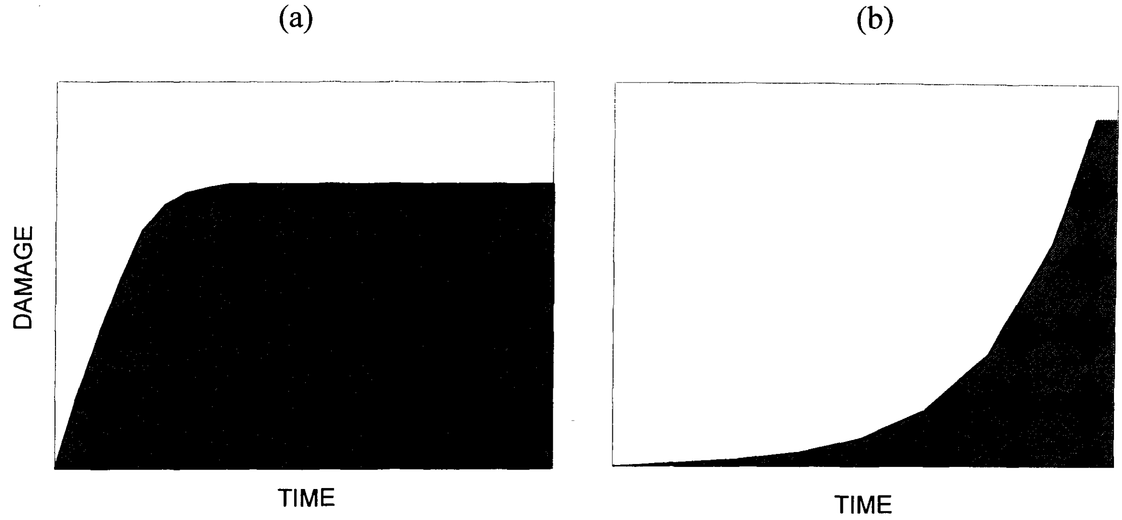

Figure 1.10. Conceptual model for crop damage resulting from (a) flooding and (b) waterlogging (from Mann and Green, 1978).

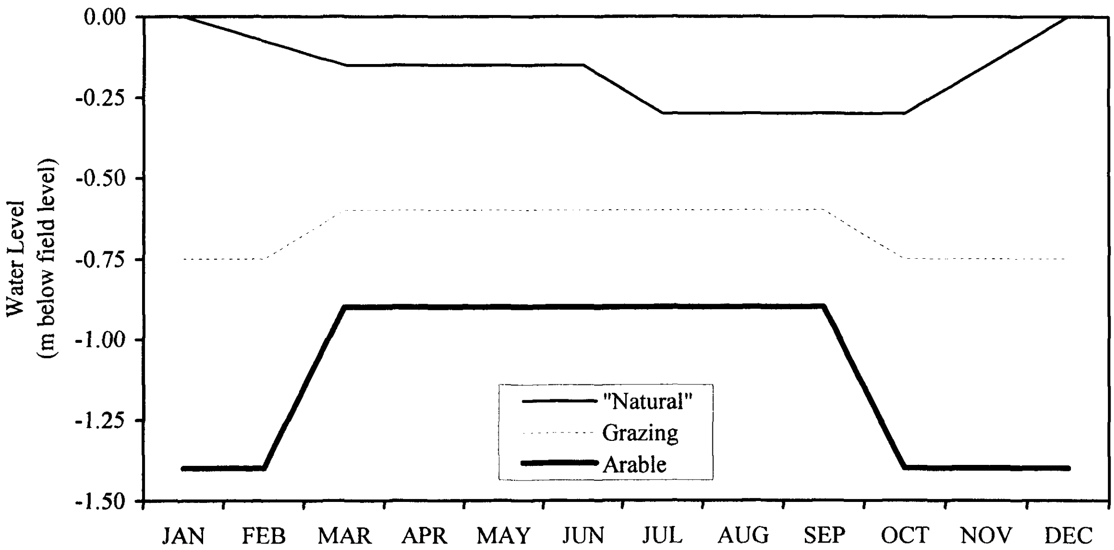

Figure 1.11. 'Natural' versus arable and grazed water level regimes for wet grassland areas (from van Bakel, 1988 and Spoor and Gowing, 1995).

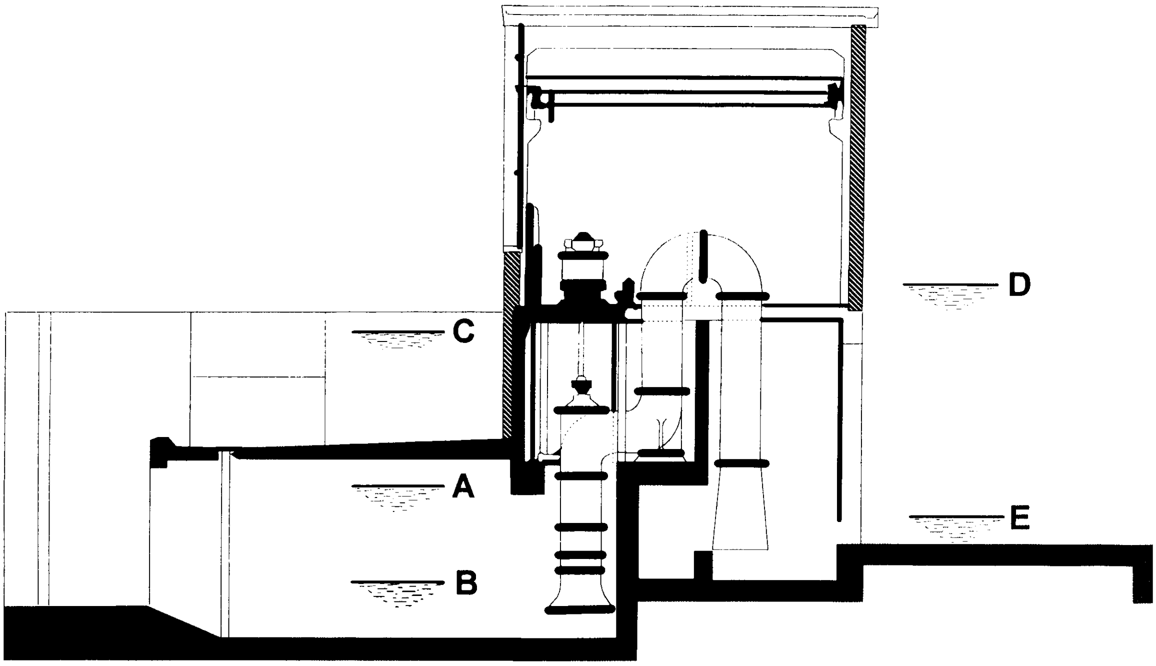

Figure 1.12. Typical arrangement of a pumping station (from Beran and Charnley, 1987). A: Pump start level, B: Pump stop level, C: Storm override level, D: summer retention level (embanked channel) and E: winter retention level (embanked channel).

		Design Flood Frequency (not more than)	
Land potential	Crops	March-November	Whole Year
Very High	Horticultural	No flood allowed	1 in 100 years
High	Root crops	1 in 25 years	1 in 10 years
Medium	Cereals	1 in 10 years	1 in 5 years
Low	Re-seeded grass	1 in 5 years	1 in 2 years
Very Low	Rough grazing land	1 in 3 years	1 in 1 years

Table 1. 11. Agricultural land drainage standards in the UK (From Shaw, 1993).

Target water levels for agricultural land in the UK can be determined based on information regarding the target crop type. The return periods employed in UK drainage design for different agricultural crops are given in Table 1.11. An important consideration where under-drains have been installed is that the water level should not exceed the outflow level (Ritzema *et al.*, 1996). Thomasson (1975) reports that drain depths are frequently between 0.7 and 1.2 m, in accordance with observations on the North Kent Marshes where under-drains are 0.7 m below the field surface (Al-Khudhairy *et al.*, 1999). For reclaimed land in the UK, under-drains are generally 0.8 m below the surface (Beran and Charnley, 1987).

The variety of objectives sought by different land use management strategies in wet grassland areas, coupled with their seasonal importance during the cropping cycle, means that flexibility is one of the most important components of the hydrological management in these areas. This has been achieved locally by the construction of sluices, weirs and other water retention structures, which depending on design, can be cheap to install. Penning board sluices, a set of planks set across the ditch to control the height of water are documented from the mid 13th Century on the Gwent Levels, where they are called 'stanks' (Rippon, 1996).

A number of sluice gates can be located strategically along the ditch system, so that water can be discharged from, or supplied into the primary ditch system to the secondary ditch. This may be a complex process, requiring the opening of one or more other sluice gates on connected ditches. Depending on the size of the ditch, the dimensions of the hydraulic structure will vary. Field scale board sluices are commonly about 4m wide (RSPB *et al.*, 1997), although on embanked channels much larger structures may be present. As part of the Ouse Washes Flood Control system for example, a structure consisting of three gates, each 7.4m wide by 6.7m deep was installed on the Hundred Foot River in 1997 at a cost of £5.2 million (ICID, 1998). Different types of structure may also be employed: on the North Kent Marshes and Gwent Levels, tidally-controlled, uni-directional valve sluices are employed on the seaward end of the wetland to drain water on the ebb tide but limiting the intrusion of saline water into the drainage system.

1.6.4. SHALLOW GROUNDWATER HYDROLOGY

Historically, the main objective of ditch water level management has been to maintain the in-field water table at levels that enable the economically viable cultivation of land. Benefits from the drainage of agricultural land accrue from the improved crop growth conditions created by drainage (earlier, higher, more dependable or better quality yields), and the improved soil workability conditions (earlier planting, more workable days and less damage to soil structure by farm machinery) (Smedema and Rycroft, 1983). The effects of waterlogging on crops are not direct, but are related to differences in soil air volume. For example, an alluvial clay soil with a water table 1.0m below the ground surface contains 6% of air volume, compared to 0-4% when the water table is 0.3m from the surface (Muller, 1992). Waterlogging impairs mineralisation and nitrification by microbes, and may cause the soil structure to disintegrate or prevent it being restored by the action of frost (Smedema and Rycroft, 1983). Prolonged waterlogging causes irreversible damage, killing the roots through the action of anaerobic bacteria (Mann and Green, 1978) and toxic products of chemical reduction.

As a result of these biochemical processes, water table depth has an especially large influence on crop productivity, a relationship that although complex, is well documented for agricultural crops. For clay soils, Cannell *et al.* (1984) has identified reductions in the yield of winter wheat of 18% associated with a water table 0.1m from the surface relative to a water table 0.9m from the surface. For a variety of cereal and grassland crops, yield as a function of water table level is given in Table 1.12, although the nature of the substrate is also a determining factor (Figure 1.13). Due to the influences on crop productivity, in farmed wet grassland areas the hydrology of shallow groundwater in wet grasslands closely reflects the use of the land. For example, for arable areas, Muller (1992) suggests an ideal springtime water table level of between 0.9 and 1.1 metres below the field surface, supported by water table measurements on the North Kent marshes, where summer water table levels 1.0 m below the ground surface were measured on arable land (Al-Khudhairy *et al.*, 1999). Higher water table levels can be maintained in wetland areas used for grazing. Cook and Moorby (1993) suggest summer water tables of between 0.3 and 0.5m below field level as ideal, closely coincident with the ideal water level requirements of Fescue (Williamson and Kriz, 1970), a common grass species of improved and unimproved wet grassland swards (see Section 1.4.).

	Water Table Depth (m below ground surface)							
Crop	0.15	0.30	0.45	0.60	0.75	0.85	1.00	1.50
Wheat	-	-	58	77	89	95	-	100
Barley	-	-	58	80	89	95	-	100
Oats	-	-	49	74	85	95	-	100
Corn	-	41	82	85	100	85	45	-
Potatoes	-	-	90	100	-	95	92	96
Mustard	52	96	100	93	95	97	99	-
Millet	41	69	80	87	98	100	93	-
Sorghum	73	86	93	100	93	-	-	-
Tall fescue	51	100	87	-	-	-	-	-
Cocksfoot	28	100	93	-	-	-	-	-

Table 1.12. Yields of agricultural crops at different water table levels. Expressed as a percentage of maximum yield (from van Schilfgaarde, 1974).

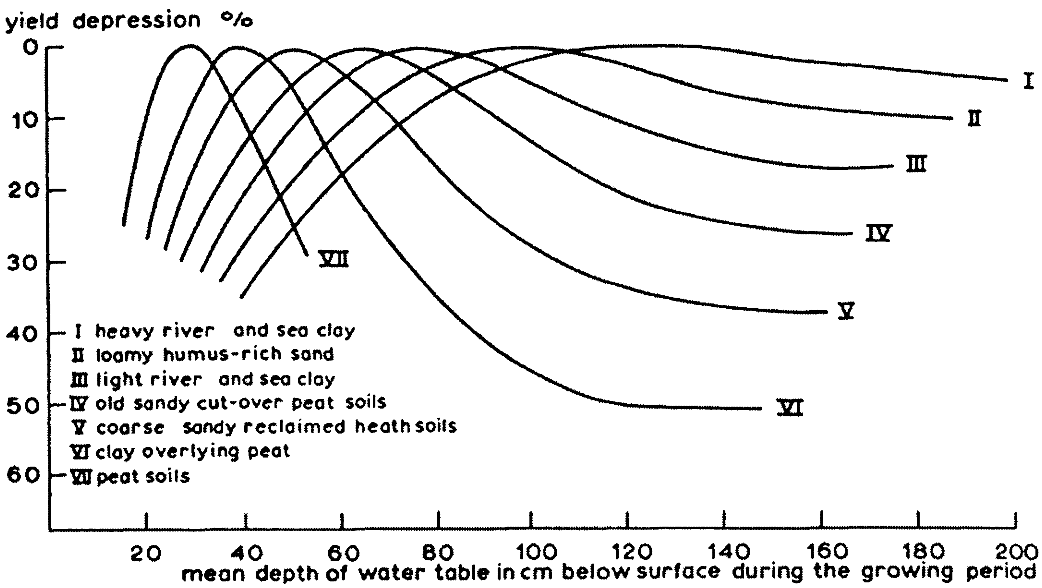

Figure 1.13. Yield depression as a function of the mean depth to the water table in different soil types (from Visser, 1958).

Unlike the hydrological regime in wet grassland ditches, where the natural phase is reversed, variations in the water table closely illustrate seasonal variations in the balance of rainfall and evapotranspiration. This reflects the difficulty of 'engineering' water table levels by varying ditch water levels, especially in areas where the hydraulic conductivity of the soil is low, such as wet grasslands with alluvial clay substrates (Section 1.5). In these areas, controlling water table height presents an added difficulty since both horizontal and vertical movement are limited (Youngs *et al.*, 1989). At certain times of year, the water table may also intercept the soil surface causing conditions of shallow flooding and may also take a long time to recover following droughty conditions. Nevertheless, drainage can reduce water table level maxima. In drained areas of the Middle Fen, Cambridgshire, Harris *et al.* (1991) have identified mean differences in water table elevations of 0.15 - 0.20m relative to un-drained areas. On the same site, the installation of a pump eliminated flooding and maintained water table levels at least 0.35m from the ground surface throughout the year, with an annual range 0.50 - 0.65m reported. Higher annual ranges, between 0.8m and 1.1m, are quoted for clay soils on the North Kent Marshes by Hollis *et al.* (1993) and the Norfolk Broads (Armstrong, 1993). On peat soils, annual ranges of approximately 0.7m have been identified by Youngs *et al.* (1989) on the Somerset Levels.

The over-riding influence of rainfall and evapotranspiration, coupled with the seasonal management of ditch water levels in intersecting ditches, gives rise to the characteristic form of the water table, which is convex in winter and concave in summer (RSPB *et al*, 1997). These shapes arise because groundwater conditions in the field centre are dominated by the influence of rainfall and evapotranspiration, whilst the influence of the ditches is greatest closest to the ditch. Cook and Williamson (1999) have measured winter freeboards (the difference between water table and ditch levels) of between 0.3 and 0.5m, with negative freeboards of an equivalent magnitude apparent in the late summer and early autumn, when ditch water level are frequently above the in-field water table. In contrast, in peat-dominated areas ditch water may contribute considerably to field water table height in the summer (Youngs *et al.*, 1989). In some areas of the Norfolk Broads the shape of the water table is essentially flat, as the prevailing hydraulic conductivities ensure the influence of the ditch system extends fully into the field (Agricultural Development Advisory Service [ADAS], 1994).

1.7. Restoration of wet grassland

Changing societal attitudes to environmental issues, coupled to recent changes in rural land use policies have provided the opportunities and economic means to address the degradation of wetland ecosystems (Pyewell *et al.*, 1994). The evolution of agricultural policy since the war has been dominated by the major objective of producing an appropriate level of farm output mainly by means of supporting farming income through pricing (Whitby, 1994). During the 1970s there was an incentive to increase arable production in wetland areas with substantial grants available for drainage and associated works (Samuels, 1993)(Section 1.3.2). By the 1980s concern had shifted towards over production and the environmental impacts of flood defence and drainage schemes, a trend that has continued during the 1990s, during which period an increasing number of statutory mechanisms for the protection and restoration of wetlands have been put in place. The word 'restore' has various nuances of meaning, and can be used to accommodate various degrees of reinstatement - repair, reconstruction, reproduction or recreation (Wheeler *et al.*, 1995). In wet grassland areas, management for restoration is mainly associated with the re-introduction of traditional agricultural practices, especially with respect to hydrological and vegetation management (Burgess and Hirons, 1990).

1.7.1. MANAGEMENT OF FIELD VEGETATION

The decline in the botanical interest of much of Britain's lowland grassland has been mainly attributable to a shift from hay making to high intensity agriculture, including the improvement and fertilisation of grassland for dairy farming (RSPB, 1994) (Section 1.3.2). Traditional management included the use of organic manure in late to early May, when grazing livestock were removed to allow the growth of the hay crop, which was cut sometime in July or August depending on weather conditions (Smith and Rushton, 1994). Because of the negative correlation between nutrient availability and species diversity in grass swards (Oomes *et al.*, 1996), restoration management of wet grassland revolves around transforming productive, species poor grasslands into less productive grasslands with a higher species density. In the Ouse Washes for example, the least grazed fields show the lowest species count (Penning-Rowsell, 1986).

Reductions in grassland productivity are achieved firstly by the cessation of fertiliser application, and secondly by reducing dry matter production by cropping (grazing or mowing), a process that reduces the net effect of enrichment (Fojt, 1992). This approach also limits the succession towards scrub and secondary woodland that would occur under naturalised conditions (Gilman, 1994). Grazing remains the simplest and least labour intensive cropping method, creating the least amount of disruption to the system. Traditional grazing practices have therefore been re-introduced in most wet grassland sites where restoration is an objective. In most cases this has not been a variation on the past land use, but simply a greater control on the density and timing of grazing. In particular the use of different species to graze the sward at different times of year can encourage not only species diversity but also structural diversity, since different grazers affect the sward in different ways (Table 1.13). In areas where ground nesting birds are present, the grazing density required to achieve the target sward may have to be adjusted to reduce the risk of nest trampling (RSPB *et al.*, 1997).

Mowing offers an alternative to grazing, allowing the wetland manager to create a mosaic of vegetation heights within the same field or land area. Indeed, mowing for silage has assumed an increasingly important role in the conserved grass output of many farming systems (RSPB, 1992). This process realises a greater food value than mowing for hay but, because mowing has to start earlier in the season, can have a negative impact on the flora and fauna of the grassland sward, including ground nesting birds. This practice is generally deployed where structural character is more important that species diversity (Wheeler *et al.*, 1995). Mowing is particularly useful for dealing with invasive plants such as Ragwort (*Senecio jacobea*) or thistles (*Cirisium vulgare*, *C. arvense*). These are 'topped' twice a year; once prior to flowering and then again a month later (Fojt, 1992). Such species tend to be unpalatable to stock, although some species exploit these types of vegetation. Managers on the Swale National Nature Reserve, North Kent, for example graze three highland sheep which feed exclusively on thistles as a means of limiting successionary processes out-competing other rarer wetland plants on the site.

Factor	Sheep	Cattle
Weight	Light - poaching unlikely	Heavy - poaching in wet areas
Appetite	Small per beast	Greater per beast
Hooves	Numerous per appetite unit - nest trampling likely	Fewer per appetite unit - nest trampling less likely
Preferred sward	Initially short	Ranker, taller vegetation
Resultant sward	Short, even	Tussocky, less even
Land Drainage	Needs to be dry to avoid disease	Able to cope with wetter conditions
Topography	Can graze steep slopes and banks	Flatter topography required, although ditches not problematic
Unit Value	Low - death less of a loss	High - death high loss
Disease	Risks are high	Risks are low
Wintering	May be outwintered, little supplementary feed needed	Over-wintering facilities required, supplementary feeding
Lookering	Intensive	Less Intensive
Routine care	Costly and intensive	Minimal
Stock retention	Fencing not robust	More robust fencing
Recent trends	Numbers increasing	Health scares have affected profitability

Table 1.13. Comparative overview of the feeding behaviour and effects on sward composition and structure of cattle and sheep (from RSPB, 1992).

1.7.2. MANAGEMENT OF AQUATIC VEGETATION

The management of vegetation in drainage ditches of wet grassland has been historically important to ensure effective drainage. Submerged and emergent plants increase the frictional resistance to flow in a channel and may block pump screens and it is therefore necessary to control these by dredging (Newbold *et al.*, 1989). Most main channels receive some treatment every year while smaller channels receive treatment on a less frequent basis: on the Monmouthshire Moors subsidiary channels are dredged every ten years (Marshall *et al.*, 1978). This type of approach has been adopted by nature conservation bodies, because in general natural wetland succession from open water to woodland carr results in a declining diversity of species (Penning-Rowsell, 1986). A cycle of cutting and clearing bank and ditch vegetation can allow ditches to have a variety of successionary communities, from those characteristic of open water, to those approaching semi-terrestrial ecosystems.

Researchers of the hydrological preferences of wetland species have identified the specific hydrological requirements of individual species, with water depth being an important factor for aquatic vegetation (Table 1.5). Based on this type of information, the appropriate conditions for target aquatic vegetation can be provided by cross-sectional re-profiling, a practice which also creates a greater diversity of micro-habitats on ditch boundaries which can be exploited by numerous biota. Re-profiling may involve the creation of a shallow shelf or berm along selected lengths, the artificial formation of riffle and pool sequences on the ditch bed, or the stabilisation of eroding banks to allow colonisation by vegetation. Newbold *et al.* (1989) provide an excellent review of the means by which this type of management can be carried out. In many cases, these ecological objectives satisfy agricultural objectives, making it a sustainable form of management in areas where grazing is the principal land use. By reducing bank angles, stock can access the ditch more easily for drinking. An added advantage is the creation of poached ditch margins favoured by some wet grassland plants and numerous invertebrates (Jones, 1992). Re-profiling can also remove spoil banks to allow ditch water to flow more easily onto fields, a practice used by the RSPB at West Sedgemoor on the Somerset Levels.

1.7.3. MANAGEMENT OF WATER LEVELS

Control of water level has tended to take priority over vegetation or animal control, which in any case may be achieved through the manipulation of a site's hydrological regime (Fojt, 1992). Past research has suggested that it is technically feasible to manipulate the processes affecting plant species assembly in order to rapidly restore wetland vegetation communities which closely resemble their semi-natural counterparts (MAFF, 1995). Where wetland degradation has been a factor of desiccation associated with the lowering of water tables, the wetting up of soils by raising ditch water levels makes them more easily penetrable for wader species which probe the ground for food (ADAS, 1994). This also provides a means of eliminating invasive plants and limiting successionary development towards scrub: drier conditions and increased availability of nutrients favours species such as *Urtica dioica*, *Epilobium hirsutum* and woody species (Fojt, 1992).

Hydrologically-based restoration management in wet grassland areas has generally attempted to redress the balance between climate and ditch water level fluctuations by recreating the 'natural' hydrological regime, possessing maxima in the winter and minima in late summer (Figure 1.11). An important component of management in winter and early spring is the promotion of surface flooding. A gradation of flooding depths and durations and of management intensities is required to maintain habitats for a variety of species (Penning-Rowsell et al., 1986). For birds, RSPB *et al.* (1997) suggest that water levels providing inundation of 30-60 % of the target site to a water depth less than 0.2 m between December and March are required, with this area being reduced to 20% in April.

For the remainder of the year, water level management should seek to reduce the incidence of surface flooding as it affects invertebrate species which provide a food source for avian species. Earthworms for example cannot withstand prolonged flooding (Newbold *et al.,* 1989), but can survive in areas where the water table is high, as 75 % of earthworm biomass is found in the top 0.05m of the soil (Voisin, 1959). Summer flooding can also have detrimental impacts on larval stages of aquatic insects. Flooding in September damages populations of non-mobile terrestrial invertebrates, and before the end of October, inundation prevents beetle and cranefly species from laying eggs (RSPB *et al.*, 1997).

1.7.4. AGRI-ENVIRONMENT SCHEMES

Hydrological manipulation is an integral component of most agri-environment schemes applied in wet grassland areas. Agri-environment schemes have their origins in the re-orientation of European agricultural policy during the 1980s, when awareness grew of the conflicts between productivist agriculture and the environment (Bartram *et al.*, 1996a). By 1985 the relative cost of dairy support alone was 32.1% of the total Common Agricultural Policy (CAP) budget, (Whitby, 1994). This led to attempts to integrate environmental protection into agricultural policy, an approach that was furthered in the Single European Act (1987), the 5th Environmental Action Plan (1992), the Maastricht Treaty (1992) and the MacSharry report (1992) (Bartram *et al.*, 1996b).

Of particular importance in this approach was Article 19 of Council Regulation 797/85 on Improving the Efficiency of Agricultural Structures, which set up a system of agri-environment schemes, which replaced previous price-support mechanisms. One of the early contentions regarded the source of funding for these schemes and as a result EC Regulation 1760/87 provided European Guidance and Guarantee Funds (FEOGA), with a maximum reimbursement of 25% (MAFF, 1989) rising to 50% in 1992, with the remainder made up by national governments. Agri-environment schemes currently represent 3% of CAP spending (Guardian 06/01/99). The UK was the first country to implement Article 19 (Wilson, 1997), which was adopted through the Agriculture Act 1986 (Section 18) (Whitby, 1994). As part of this Act, the Minister of Agriculture was required to

'seek to achieve a reasonable balance between the maintenance of a stable and efficient agricultural industry, the economic and social interests of rural areas, the conservation and enhancement of the natural beauty of the countryside (including its fauna, flora, geological and physiographical features) and the promotion of its enjoyment by the public.'

Section 18 of the Act gave the minister powers to designate areas with special standards of protection, from which agri-environment schemes emerged.

There are currently at least eight different agri-environment schemes operating in the UK, offering payments of between £8 to £500 ha^{-1} yr^{-1} depending on the management practices adopted. Agri-Environment schemes represent a major political and financial commitment to the conservation of wet grassland areas at the national level, and the amount of funding devoted to the scheme increased steadily during the 1990s. Prescriptions cover a range of daily and seasonal farming techniques and include restrictions and prohibitions (drainage works, fertiliser use, grazing levels) and some positive works (maintenance of hedges, barns, ponds) (MAFF, 1989). Two of these schemes, the Environmentally Sensitive Area (ESA) and Countryside Stewardship (CS) schemes, also have specific prescriptions relating to the control of ditch water levels in the areas to which they are applied.

1.7.4.1. The Environmentally Sensitive Area Scheme

In the UK, adoption of EC Regulation 797/85 followed environmental concern about the effects of agricultural intensification, and resulted in the introduction of the ESA concept. In 1985, in partnership with the Countryside Commission, MAFF set up a scheme called the Broads Grazing Marshes Conservation Scheme (BGMCS) (MAFF, 1994), which by providing payments for the reversion to extensive pastoral farming, was widely considered as a 'pilot' for future ESA projects. The BGMCS attracted 90% of the farmers in the target area of the Norfolk Broads. Following its success, the first round of ESAs was launched in 1987, followed by further rounds in 1988, 1993 and 1994 (Table 1.14). As a result, the area covered by agreements increased dramatically (Figure 1.14.) as did expenditure on ESA schemes, from £8.3mn in 1988/89 (MAFF, 1989) to £43 m in 1995 (Morley, 1993). By 1996/1997 financial provision for ESAs was £50.6 million (Bartram *et al.*, 1996b).

There are currently 22 ESAs in England covering 3,376,500 hectares, 10% of all agricultural land (Figure 1.14). A number of these are, or contain, wet grassland habitat (Table 1.14). All ESAs are designated in order to promote land-use activities sympathetic to the conservation interest (Fojt, 1992). Only farmers within the boundary of the ESA can enter the 10 year management agreements, with annual payments of between £8 and £500 ha^{-1} yr^{-1} depending on the management practices adopted that are incorporated as a series of management tiers (Table 1.15). An important distinction relates to the water levels maintained in the ditch system: the maintenance of higher water levels is rewarded with higher levels of payment.

Year	Site Name
1987	Pennine Dales, *Norfolk Broads*, *Somerset Levels and Moors*, South Downs East, West Penwith
1988	Breckland, North Peak, Shropshire Borders, *Suffolk River Valleys*, South Downs West, *Test Valley*
1993	*Avon Valley*, Exmoor, Lake District, *North Kent Marshes*, South Wessex Downs, South West Peak
1994	Blackdown Hills, Cotswold Hills, Dartmoor, *Essex Coast*, Shropshire Hills, *Upper Thames Tributaries*

Table 1.14. Areas notified in the UK as Environmentally Sensitive Areas (ESA) between 1987 and 1994 (from a variety of sources). Areas shown in italics are mainly wet grassland habitat or include extensive tracts of land of this character.

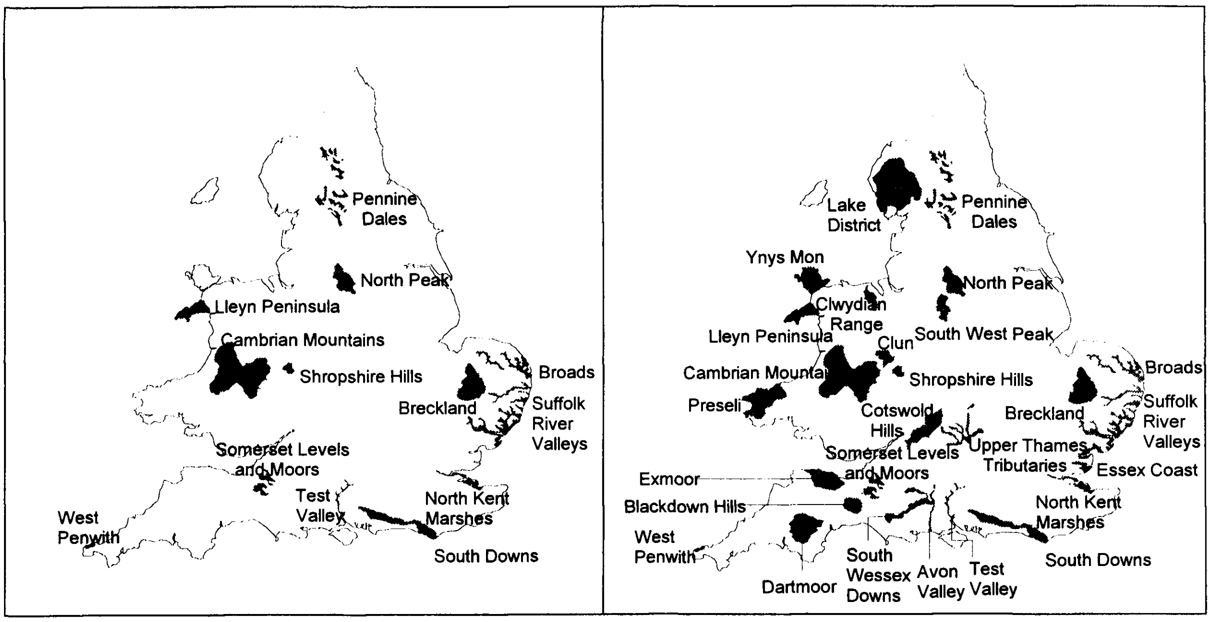

Figure 1.14. Environmentally Sensitive Areas notified in England and Wales in (a) 1988 and (b) 1994 (from Whitby, 1994).

TIER 1. To maintain the Somerset Levels and Moors landscape and grassland.

Maintain grassland and do not plough, level or re-seed the land. You many use a chain harrow or roller but no other form of cultivation is allowed.

- Graze with cattle or sheep but avoid poaching, undergrazing or overgrazing.
- If you cut the grass for hay or silage, graze the aftermath.
- Do not exceed your existing level of inorganic fertiliser and in any case do not exceed 75 kg of nitrogen, 37.5 kg of phosphate and 37.5 kg of potash per hectare (60 units of nitrogen, 30 units of phosphate and 30 units of potash per acre) each year. Do not use fungicides or insecticides.
- Do not apply herbicides except to control creeping buttercup, soft rush, nettles, spear thistle, creeping or field thistle, curled dock, broad-leaved dock or ragwort. Apply herbicides by weed wiper or spot treatment.
- Do not apply lime, slag or any other substance to reduce soil acidity.
- Do not install under-drainage, do not mole drain, and do not subsoil or tunnel plough. Do not substantially modify your existing drainage system.
- Maintain existing field gutter, surface piping, rig and furrow, ditches or rhynes by mechanical means. Do not install additional surface piping.
- Do not spray irrigate your land.
- Maintain hedges, tree and pollarded willows in accordance with local custom.
- Do not replant any additional trees nor allow natural establishment of additional trees/bushes.
- Do not damage or destroy any feature of historic interest.
- Obtain written advice on siting and materials before constructing buildings, roads or any other engineering operations which do not require planning permission.

Maintain existing gates with fencing. Do not erect any additional fencing.

From 1 April to 31 October maintained at or above the penning level, provided since 1987, by the relevant IDB and not more than 45 cm below mean field level and from 1 November to 31 March, maintained at or above the winter level provided since 1987 by the relevant IDB with at least 15 cm of water in the bottom of the ditches.

Or, to obtain a supplementary payment of £70 per hectare:

- From 1 May to 30 November maintained at not more than 30 cm (12") below mean field level and from 1 December to 30 April maintained at not less than mean field level so as to cause conditions of surface splashing.
- Agreement holders must not pump below these levels which will be fixed by reference to gauge boards set to Ordnance Datum Newlyn.

Table 1.15. Management guidelines for the Somerset Levels and Moors Environmentally Sensitive Areas (ESA) scheme (MAFF, 1995).

TIER 2. To enhance the ecological interest of grassland.
• Do not use chain harrow or roller between 31 March and 1 July • Do not exceed your existing level of inorganic fertiliser and in any case do not exceed 25 kg of nitrogen, 12.5 kg of phosphate and 12.5 kg of potash per hectare (20 units of nitrogen, 10 units of phosphate and 10 units of potash per acre) each year. • Unless traditionally the land has been used just for grazing each year mow at least one third (or one year in three) of the land but not before 1 July and do not graze the land prior to laying it up. • Do not cut or top the grass after 31 August. • Do not graze with sheep from 1 September to 1 march • Do not use herbicides to control creeping buttercup. • Water levels in ditches and rhynes must either: From 1 April to 31 October maintained at or above the penning level, provided since 1987, by the relevant IDB and not more than 45 cm below mean field level and from 1 November to 31 March, maintained at or above the winter level provided since 1987 by the relevant IDB with at least 30 cm of water in the bottom of the ditches. Or, to obtain a supplementary payment of £70 per hectare: - From 1 May to 30 November maintained at not more than 30 cm below mean field level and from 1 December to 30 April maintained at not less than mean field level to cause conditions of surface splashing. - Agreement holders must not pump below these levels which will be fixed by reference to gauge boards set to Ordnance Datum Newlyn.
TIER 3 To further enhance the ecological interest of grassland by the creation of wet winter and spring conditions on the Moors.
• Do not carry out mechanical operations between 31 March and 1 July. • Apply no inorganic fertiliser and do not exceed your existing level of organic manure provided it is only home produced cattle farmyard manure and does not exceed 25 tonnes per hectare (10 tons per acre) per annum. No slurry should be applied. • Graze only with cattle but do not graze before 20 May in any year. Do not exceed a grazing density of one animal per 0.75 hectare (one animal per 1.8 acres) from 20 May to 8 July. Do not cause poaching, over-grazing or under-grazing. • Do not make silage. Unless traditionally the land has been used just for grazing each year mow at least one third (or one year in three) of the land but not before 8 July and do not graze the land prior to laying it up. • Do not cut or top the grass after 31 August. • Do not use herbicides to control creeping buttercup. • Water levels in ditches and rhynes must: From 1st December to 30th April be maintained at than mean field level to cause conditions of surface splashing. **Public Access Tier** Payments are also available for creating new public access for walking and other quiet recreation.

Table 1.15. Continued

1.7.4.2. The Countryside Stewardship Scheme

Countryside Stewardship (CS) is the second largest agri-environment scheme in the UK (Table 1.16). In 1999 there were 8600 agreement holders, an increase of 65% from 1996/97 signatories, and CS covered an area 152,000 ha in extent (MAFF, 1999). By combining conservation, access to the countryside with commercial land management through a national system of incentives (Countryside Commission, 1991a), it is less concerned with the regulation of agricultural practices, which is the main objective of the ESA scheme. It is specifically targeted to protecting and enhancing the nature conservation interest of a number of key semi-natural habitat types in England, the distribution of which has declined dramatically (Fojt, 1992). Target habitats include chalk and limestone grassland, lowland heath, watersides, coasts, uplands, historic landscapes, traditional orchards, old meadows and pastures and traditional field boundaries (Countryside Commission, 1991a). The main objectives of CS are to

- conserve landscapes and views,
- improve and extend habitats for plants and animals,
- preserve archaeological and historic features,
- provide new opportunities for people to enjoy the countryside,
- restore neglected land and
- create new wildlife habitats and landscape features (MAFF, 1999).

Of particular bearing to wet grasslands is the Water Fringes option of CS, which aims to

- support and re-introduce traditional management to sustain and extend meadows and pastures and associated wildlife,
- restore and protect characteristic waterside features, and
- for existing areas of traditionally managed land, select arable land that would link the fragmented remnants of existing pastures and meadows.

There are two 'tiers' to this option of the scheme and annual payments for CS range from £15 to £280 per hectare depending on the management adopted (Countryside Commission 1991b) (Table 1.17). The management prescriptions associated with both schemes include intervention with both the hydrology and grazing regime of the target area (Table 1.18). Tier 1 relates to the *maintenance* of the existing grassland, Tier 2 focuses on the *re-creation* of waterside grassland on arable or ley grassland. Tier 2 differs from Tier 1 only in that in areas of previously arable land or improved grassland, some reseeding may be required. Following this operation, prescriptions are equivalent to those for Tier 1, and are subject to 10-year management agreements, as in the case of ESAs.

Expenditure on the CS scheme rose progressively during the late 1990s. This has been mainly related to the success of agri-environment schemes in general, with increases in the area of fen, marsh and swamp of 27% and an increase of 38% in the plant diversity around around fields (Guardian 30/11/2000). Due to its wider habitat remit and wider applicability in the context of the broader rural economy, the CS scheme is currently the main 'green grant' scheme of the English government with £500 million allocated to the scheme between 2000 and 2006 (MAFF, 2000a). This extra funding has been targeted towards new agreements, but also to increase the current levels of subsidy for 'environmentally-friendly' farming practices.

Scheme	Expenditure 1996/97 (£mn)	No. of agreements
Environmentally Sensitive Area	55.0	7700
Countryside Stewardship	12.2	5200
Habitats Scheme	3.5	low
Nitrate Sensitive Area	6.1	Not applicable
Countryside Access	3.0	low
Moorland Scheme	5.3	low
Organic Aid Scheme	1.2	800
Tir Cymen	5.0	556

Table 1.16. Expenditure on agri-environment schemes in the England and Wales in 1996/97 and number of agreements in place in 1996/1997 (from Bartram *et al.*, 1996a, 1996b).

Code	Management Targets	£/ha
R1	*Tier 1:* management of existing permanent grassland	£70/annum
R2	*Tier 2:* re-creation of traditional waterside landscapes on arable land or ley grassland	£225/annum
A	Land made available for public access	£50/annum
Supplementary payments		
r	Tier 1 land for initial work needed to establish or re-introduce grazing	£40 1st year payment
r	Tier 2 land for additional work to help re-create traditional waterside grassland	£40 1st year payment
W	The creation of waterside features such as reedbeds, fens and carr	£40/annum

Table 1.17. Payments provided by the Countryside Stewardship scheme (from Countryside Commission, 1991b).

Prescriptions relating to grassland management
• grassland should be managed by light grazing of cattle for at least 10 weeks in each year, or by cutting for hay • stocking rates not greater than 6 ewes/1.5 cattle per ha. Lower between 1 March -30 June, as a guide no more than 4 ewes/1 steer or Heifer per ha • no pesticides
Prescriptions relating to the management of water levels and aquatic vegetation
• no new drainage • summer water levels maintained at levels associated with traditional grassland management Guide levels = April to October –20cm, Winter (October-March) 0cm from bank level • ditches should be maintained in a 5-10 year rotation without the use of herbicides

Table 1.18. Management guidelines for the Countryside Stewardship scheme (from Countryside Commission, 1991b).

1.7.5. WATER LEVEL MANAGEMENT PLANS

Perhaps one of the most significant developments in recent years concerning the hydrological management of wet grasslands is the water level management planning initiative (Swash, 1998). Water Level Management Plans (WLMPs) detail how water levels within a defined area can be managed to balance the requirements of a range of activities, including agriculture, flood defence and conservation. Guidance on preparing WLMPs were first issued in the publication of *'Water Level management Plans – a guide for operating authorities'* (MAFF, 1994). This publication states that Plans should be produced for areas where some form of water level management is already in place, with the highest priority afforded to internationally important sites, such as Sites of Special Scientific Interest (SSSIs) which qualify as candidate Special Areas of Conservation (cSACs) or Special Protection Areas (SPAs).

The production of the Plans involve all those whose interests may be affected within the area covered by the WLMP (MAFF, 1994). WLMPs differ from agri-environment schemes (Section 1.7.3) in that they are a statutory responsibility of the Internal Drainage Board, normally the Environment Agency (EA). Unlike agri-environment schemes, they are not voluntary and do not provide landowners with a subsidy for the losses incurred by retaining higher water levels. Neither are WLMPs associated with financial support for the formulation and implementation of the schemes. As a result, whereas 560 SSSIs were identified in 1994 as areas that would benefit from the sensitive management of water levels, by the end of 1998 only 310 plans had been completed (MAFF, 1999b). In recent times, grant aid has been made available to address these problems. The publication entitled '*Water Level Management Plans: Additional guidance notes for operating* authorities' (MAFF, 1999b) has set out a scheme for funding the capital costs associated with these schemes. In SPAs or Ramsar sites, a figure equivalent of £300 $ha^{-1}yr^{-1}$ is employed based on calculations of the scheme life duration (MAFF 1999b). More limited funding (£175 $ha^{-1}yr^{-1}$) is available where SSSI status is the only designation. Nevertheless, the continued lack of subsidies has meant that even in areas where WLMPs have been undertaken since 1998, few have been implemented on the ground. A particular problem is that compromise water levels which satisfy agriculture and nature conservation are difficult to establish, although some ditch water level regimes have been proposed (Figure 1.15).

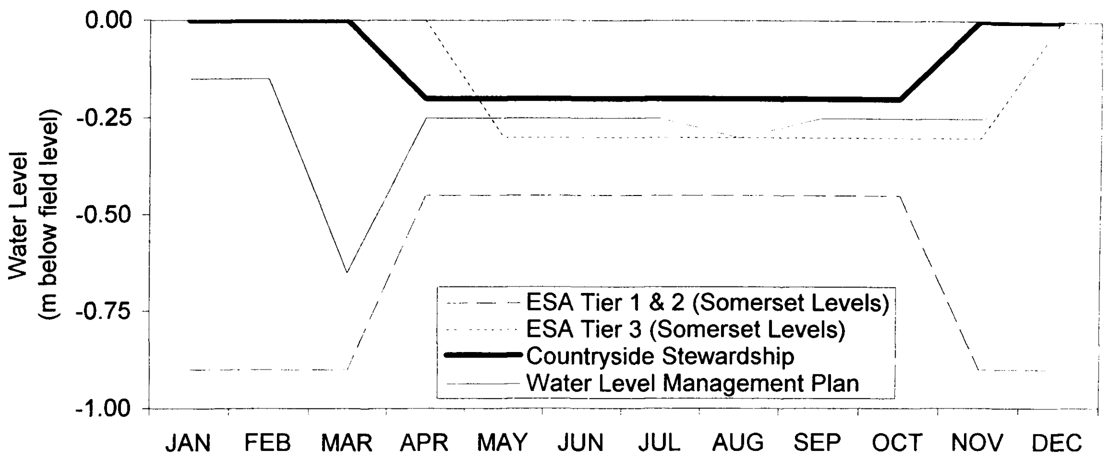

Figure 1.15. Target ditch water level regimes associated with various wet grassland restoration strategies (from a variety of sources).

Key environmental legislation relevant to inland flood defence works in England and Wales
• Wildlife and Countryside Act, 1981 • Water Resources Act, 1991, Sections 2(2), 16 and 17 • Land Drainage Act, 1991, Sections 12 and 13 • Land Drainage Improvement Works (Assessment of Environmental Effects) • Regulations ST 1988 No 1217 • Town and Country Planning (Assessment of Environmental Effects) Regulations SI 1988 No1999 • Town and Country Planning (Listed Buildings and Conservation) Act, 1990 • Ancient Monuments and Archaeological Areas Act, 1979
European Union Directives which are relevant to the environmental aspects of inland flood defence works
• Council Directive 79/409/FEC on the conservation of wild birds • Council Directive 85/337/EEC on the assessment of the environmental effects of public and private projects on the environment • Council Directive 92/43/EEC on the conservation of natural habitats and of wild fauna and flora (Under this legislation, Member States are required to list potential Special Areas of Conservation).

Table 1.19. Environmental legislation and European Directives Relevant to Inland Flood Defence Works in England and Wales (from Crofts and Jefferson, 1994).

1.7.6. LEGISLATION

Schemes that directly address the restoration of wet grassland habitats have been coupled with the revision of numerous Acts of Parliament thought to be detrimental to the successful protection of the quality and extent of wet grassland habitats. An overview of existing legislation of relevance to wet grasslands is provided in Table 1.19. The variety of institutions involved within this legislation, incorporating governmental, local and independent authorities, illustrates the institutional complexity which characterises the protection and management of wet grasslands, a feature shared with wetland management networks worldwide (Hollis, 1994). At the international level, the UK is required to protect wet grasslands of international importance under the auspices of the Ramsar Convention (1972). It is also bound by certain European Union (EU) Directives. The EU Habitats Directive 92/43/EEC (Conservation of Natural Habitats, Flora and Fauna) requires it to take measures to maintain or restore habitats with a favorable conservation status and species listed in the Annexes to the Directive.

Numerous legislative forms of protection exist in the UK (Table 1.20). Protection of wet grassland sites in the UK has traditionally been achieved by the notification of important areas as SSSIs and in 1993 there were 175 SSSIs containing significant wet grassland habitat (Denny, 1993). The concept of the SSSI was introduced under the 1949 National Parks and Access to the Countryside Act. These protected sites currently make up about 8 % of the British countryside. The main objective of SSSI notification is the control of industrial, urban or agricultural development. The specific basis for protection of SSSIs was furthered throughout the 1980s and 1990. The Wildlife and Countryside Act 1981 applied very positive nature conservation policies to the drainage authorities in England and Wales, providing greater protection and encouraging countrywide ecological surveys (RSPB, 1994). On the basis of rarity, each species was issued with a special standard of protection, with rarity determined as a function of the number of 10x10 km squares in which the species appeared on a national grid. These data have been published in the form of British Red Data Books (Bratton, 1991) and have provided a means of examining the total area and geographical distribution of major habitat types on a national basis and target specific areas for protection.

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Under the Water Act of 1989 drainage authorities were given special duties to further conservation because of the damage to habitats that could be caused by drainage operations. These duties were specifically set out in an illustrated booklet, *'Conservation Guidelines for Drainage Authorities'* (MAFF, 1988). Land Drainage Improvement Works (Assessment of Environmental Effects) Regulations were issued in 1988 and included the appointment of environmental representatives to Regional Flood defence committees (RSPB, 1995). The formulation of a new Land Drainage Act in September 1994 gave drainage boards the duty to further the conservation of wildlife when making decisions relating to flood defence and land drainage and empowered ministers to intervene to prevent drainage activities proposed by IDBs which were likely to damage nature conservation interests of national and international importance. This restructuring of the institutional framework for environmental protection in the UK was completed by the creation of the Environment Agency (EA) in the UK on the 1st April 1996, which assumed the land drainage function previously exercised by the National Rivers Authority (NRA).

1.8. Problems Facing Wet Grassland Restoration

1.8.1. EXISTING LEGISLATION

The direct loss of wet grasslands by drainage and conversion to arable has now largely ceased (RSPB, 1996), although it is difficult to find a wetland site of any significance that is not under either direct pressure from development or subject to threat from activities on its periphery (Fojt, 1992). Indeed, wetlands in the UK are under greater threat than ever (Denny, 1993), although current concern is related to the decline in ecological quality, rather than changes in extent. Indeed, the generally positive conclusions regarding the increases in extent of habitats such as marsh, swamp, fen and hedgerows in the UK during the 1990s contrasts with the marked declines in the quality of grassland, downland and bogs apparent during the equivalent period (Guardian, 30/11/2000). However, the rate and scale of damage to wetland SSSIs for example, has proved beyond doubt that present protective legislation mechanisms are inadequate (RSPB, 1996). For example, much of the decline of the extent of the Gwent Levels shown in Figure 1.2 has occurred since the site's designation as a SSSI, awarded in the 1950s. Lord Mustill, sitting on a case relating to illegal drainage works on a wetland SSSI stated that

' it needs only a moment to see that this [SSSI] regime is toothless. Within months the owner will be free to disregard any notification and carry out drainage operations... the Act does no more than give the Council [NCC] a breathing space within which to apply moral pressure with a view to persuading the owner to make a voluntary agreement' (RSPB, 1994).

Agri-environment schemes have also received criticism. Any scheme should have environmental objectives and performance indicators designed to test the objectives: without these it is impossible to determine whether real conservation benefits have been achieved (Bartram *et al.*, 1996). Indeed, studies on the environmental impacts of the ESA scheme have been sparse. Studies conducted have been inconclusive about the ecological success of the scheme, which in any case has been proclaimed successful, most probably due to political pressures at the European level (Wilson, 1997). A review of the extensification premium for livestock has also been argued (Bartram *et al.*, 1996). Landowners have consistently highlighted that current payments are insufficient to address losses in yield incurred by hydrological management for wildlife. Flooding is widely used in grassland agriculture, but excessive application of uncontrolled irrigation results in the development of low yielding vegetation, such as sedges and rushes (Sprague, 1959). Higher water levels also have an effect on land accessibility at crucial times of year and limit the movement of machinery around fields (Muller, 1992). Some of these issues have been at least partially resolved in recent times by including subsidies within the schemes to, for example, tackle rushes in areas where high water levels have been maintained.

Mechanism	Enabling Legislation	Organisation(s) Involved
Site of Special Scientific Interest (SSSI)	S.28 1981 Wildlife and Countryside Act (1985 Amendment)	English Nature, Scottish Natural Heritage (SNH), Countryside Council for Wales (CCW)
Areas of Special Scientific Interest (ASSI)	Pail IV 1985 Nature Conservation and Amenity Lands (Northern Ireland) Order (1989 Amendment)	Department of the Environment for Northern Ireland (DOENI)
National Nature Reserve (NNR)	S.19 1949 National Parks & Access to the Countryside Act S.35 1981Wildlife & Countryside Act	NCCE, SNH, CCW
Natural Heritage Area	S.6 Natural Heritage (Scotland) Act 1991	SNH, Scottish Office, CCW
Local Nature Reserve	S.21 1949 National Parks & Access to the Countryside Act	Local Planning Authorities, NCCE, SNH, CCW
Management Agreement (S.39)	S.39 1981 Wildlife & Countryside Act	Rural Local Planning Authorities including National Park Authorities
Area of Special Protection (Statutory bird sanctuary)	S.3 1981 Wildlife & Countryside Act under 1954 Protection of Birds Act	Department of the Environment (DoE)
Wetland of International Importance (Ramsar site)	Ramsar Convention on Wetlands of International Importance, especially as Waterfowl Habitats (Iran, 1971)	DoE, Welsh Office, Scottish Office, NCCE, CCW, SNH
Special Protection Area (SPA)	Article 4 EC Directive (EEC/79/409) on the Conservation of Wild Birds	DoE, Welsh Off-ice, Scottish Office, NCCE, CCW, SNH, JNCC
Special Area for Conservation (SAC)	Article 7 of the EC Habitats & Species Directive	DoE, Welsh Off-ice, Scottish Office, NCCE, CCW, SNH, JNCC
Environmentally Sensitive Areas (ESA)	S.18 1986 Agriculture Act	MAFF, WOAD, SOAFD, DoE, EN, CCW,DOENI, Countryside Commission
Countryside Stewardship	N/A	Countryside Commission
Tir Cymen	N/A	CCW
NGO Nature Reserve	N/A	The Wildlife Trusts, RSPB, National Trust

Table 1.20. Protection and enhancement mechanisms for lowland wet grassland in the UK.

1.8.2. THE 'SCIENCE BASE' FOR WATER LEVEL MANAGEMENT

For the protection and restoration of wetland sites therefore, sound management strategies have to be developed (Denny, 1993). Changes to field flora and the workability of the land associated with higher water levels can generally be predicted prior to the implementation of the scheme in areas where hydrological monitoring networks are in place. When coupled with information describing the physical characteristics of the target catchment (*e.g.* field elevations and soil properties), hydrological data such as ditch water and water table levels can be employed to identify areas within the target catchment where farming practices are most likely to be detrimentally affected by higher water levels. A large part of landscape ecology also coincides with the domain of interest of hydrology (Kundzewicz *et al.*, 1991). Hydrology provides a seasonally variable template against which wetland plant communities can develop (Gilman, 1994), and is the single most important determinant of the establishment and maintenance of specific types of wetlands and wetlands processes (Mitsch and Gosselink, 1986). The basis of ecologically sound management is therefore a clear understanding of the hydrological system and all its aspects (van Diggelen *et al.*, 1991; Spellerberg *et al.*, 1991; Reed, 1993; Maltby, 1996; Hollis and Thompson, 1998, Thompson *et al.*, In press).

Current knowledge of wetland functions and hydrology however, has tended to prove inadequate for the development of wetland management prescriptions that will give predictable results from economically viable systems of management (MAFF, 1995). This is supported by Maltby (1996), who states that '*the science base is still inadequate in explaining how wetland ecosystems work and how environmental factors and processes interact to control functioning*'. In the context of wet grassland habitats, this can be ascribed to the limited understanding of wetland hydrology (Lloyd *et al.*, 1993; Gilman, 1994; Denny, 1993; Cook and Moorby, 1993; Section 1.1). This aspect complicates the evaluation of the impacts of changes to the management *status quo* on natural and agricultural systems, mainly because baseline data describing the restoration ideal, the 'natural' system, are not available (Denny, 1993). Similarly, the paucity of information limits the wetland manager's ability to predict the impacts of any future management options or climatic conditions.

This is the template within which current policies and methods for wet grassland restoration in the UK operate. Policies such as agri-environment schemes or Water level Management Plans have a strongly hydrological focus, as highlighted in Section 1.7. However, the effectiveness of manipulating ditch water levels is largely unknown in hydrological or ecological terms (Armstrong, 1993) and the amount of water that can be moved from conventional ditches to the field centre may in any case be insufficient to maintain high water table levels in the summer months (Gilman, 1994; Gavin, 2001). Higher ditch water levels imply a greater use of water to be delivered through the main water courses so that water supply may therefore be a limitation in any venture that aims to control soil water regime (Youngs *et al.*, 1991). This is further highlighted by the continuing difficulty provided by the design of water level regimes that reconcile the interest of nature conservation and agriculture on the two banks of the same ditch. Apart from being of paramount importance for the successful restoration and recreation of truly 'natural' wetland ecosystems, these issues are of economic importance: considerable funds are devoted to agri-environment schemes in England and Wales and there is considerable need to evaluate the cost and benefit of these to the public purse.

Hydrological simulation models offer the potential to address all these issues, reconstructing both past hydrological conditions and predicting the effects of future management strategies on diverse wetland stakeholders prior to their implementation (Al-Khudhairy et al., 1999). Models do however have the disadvantage of requiring an extensive programme of data collection that impose considerable costs and staff time on wetland managers. Problems are compounded by the fact that much less information is available on the instrumentation, and therefore on the hydrology of, flat low-lying catchments than areas of high relief (Marshall, 1993).

In the context of wet grassland areas, data collected for modelling purposes will necessarily include descriptions of the morphology and management of the drainage system, the physical characteristics of the soils and topographical information. For the purpose of the calibration and verification of the model, data describing the component modeled will also be a requirement. Rainfall and evapotranspiration (ET) data are also a pre-requisite, although in the case of the latter an important limitation is the difficulty of calculating or measuring ET directly (Souch *et al.*, 1996). Application of the empirical equations frequently used becomes increasingly difficult as the watershed departs from the characteristics of agricultural land, which generally has simple topography and homogenous ground cover (Claassen and Halm, 1996). Conditions evident in many wetlands, where shallow inundation creates a 'patchwork' sward, therefore cannot be represented by traditional methods as actual evapotranspiration may not have a consistent relationship to either calculated potential evapotranspiration or water table depth (Bradley and Gilvear, 2000; Gavin, 2001).

Few catchment-based, operational hydrological models can therefore be reported for lowland areas in the UK. Hydrological models available have generally been applied to upland watersheds where hydrology is less dependent on water table level and water surface storage (Giraud *et al.*, 1997). In wet grasslands, most modeling studies have been limited to field scale studies, focusing primarily on water table variations. These water table models include those proposed by Armstrong (1993) and Youngs *et al.* (1989) that have illustrated the value of integrated modeling studies to evaluate the impact of total management schemes (MAFF, 1995).

This thesis aims to address some of the recurring themes associated with the relationship between wetland management strategies described in this Chapter and scientific hydrology. The thesis is primarily concerned with the collation, collection and analysis of hydrological data to inform wetland hydrological management on the Pevensey Levels wetland, East Sussex, England. A central component is the application of the water balance approach (Novitski, 1978) at a variety of spatial and temporal scales. All subsequent Chapters deal with some aspect of this approach. In this thesis, the water balance approach is employed to address the sustainability of various management options relative to wetland stakeholders. In doing so, it seeks to illustrate the value of hydrological data and their application within hydrological models to inform decisions regarding wetland management strategies. Two spatially distinct modelling studies are presented, dealing with the catchment- and field-scale hydrology of the wetland respectively. In the context of the modeling studies presented, an evaluation of the minimum data requirements of modeling exercises and water balance assessments in wet grassland areas is implicit in the analysis.

The hydrological models that have been developed as part of this thesis, and the approaches that have been applied within them, are employed primarily to address issues related to the hydrologically-based restoration schemes that are either in operation, or have been proposed, for this wetland site. Many of these schemes are equivalent to those considered in Section 1.7. Others are site-specific in nature. A detailed description of the contents of this thesis and the rationale for the thesis structure is provided in Section 2.8.4. This rationale is set out following a discussion of the historical and current hydrological and water resource management on the Pevensey Levels wetland in Chapter 2. Chapter 2 describes the knowledge of hydrological functioning prior to the initiation of this thesis, identifies key aspects of land use and its influence on the control of local hydrology, and highlights the socio-economic issues associated with the restoration of the site. This discussion identifies the crucial importance of hydrology in all aspects of the management of the Pevensey Levels wetland, providing support for the integrated hydrological studies presented in later sections of this thesis.

CHAPTER 2

THE PEVENSEY LEVELS WETLAND:
A DESCRIPTIVE HYDROLOGY

2.1 Introduction

The Pevensey Levels, an area of lowland wet grassland located between Eastbourne and Bexhill-On-Sea in East Sussex, England (Figure 2.1), share many of the features associated with wet grasslands described in Chapter 1. The wetland has been progressively reclaimed from the sea since the Middle Ages, and has undergone the transformation from salt marsh to fresh water marsh by enclosure behind seawalls and the construction of drainage ditches to evacuate flood waters more effectively. The result is a flat landscape, dominated by an intricate network of ditches which bound fields on all sides, a characteristic feature of the morphology of wet grassland habitats (Section 1.6.1). The ditch network is subject to intensive hydrological management at all scales to satisfy agricultural objectives in the area (Glading, 1986), a practice which is generally perceived to have a negative influence on the nature conservation value of wet grassland in the UK (Section 1.2.3).

The Pevensey Levels provide a characteristic example of the anthropogenic forces commonly working against inherent hydro-ecological processes in wetlands. Reclamation and the instatement of minor drainage measures on the Pevensey Levels wetland until 1900 not only contributed to the agricultural productivity of the marshland landscape, but also created a variety of aquatic, semi-aquatic and terrestrial ecosystems exploited by a variety of flora and fauna of national importance in nature conservation terms. The Levels are therefore an excellent example of a 'cultural landscape', as defined by Spellerberg *et al.*(1991) and discussed in Section 1.1. Throughout the 20th Century, drainage has allowed the optimisation of the timing and intensity of farming operations, with a perceived negative effect on the biodiversity value of the wetland. In recent times, ecological decline has been addressed by the introduction of a series of hydrologically-based wetland restoration strategies equivalent to those considered in Section 1.7.4.

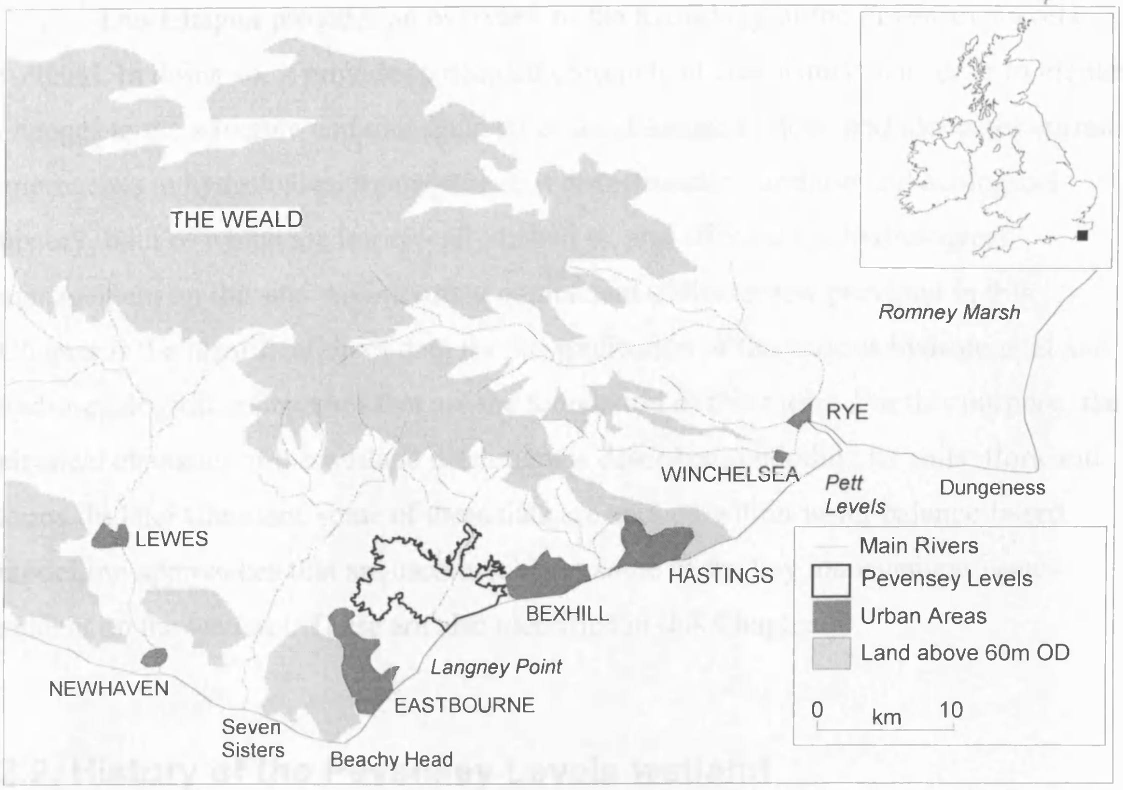


Figure 2.1. Location map of the Pevensey Levels wetland (from Jennings and Smythe, 1985).

The hydrological focus evident within wetland restoration schemes on the Pevensey Levels and in UK wet grasslands in general illustrate the importance of hydrological studies to address wetland management concerns. This thesis is primarily involved in the provision of tools, and hydrological assessments, that may provide the scientific basis for current and future water level management strategies on the Pevensey Levels wetland. In contrast to the situation in other wet grassland sites, the hydrology of the Pevensey Levels wetland has been previously considered by a number of authors, including governmental and non-governmental organisations involved in the management of the wetland. On the Pevensey Levels, these authorities can be crudely sub-divided into those with interest in agriculture (local landowners), flood defence (the Environment Agency [EA], previously the National Rivers Authority [NRA]) and nature conservation (English Nature [EN] and the Sussex Wildlife Trust [SWT]). A considerable volume of hydrological data is also available but is subject to some of the problems identified by Beran (1982) (Section 1.1).

This Chapter provides an overview of the hydrology of the Pevensey Levels wetland. In doing so, it provides a detailed chronicle of site history that seeks to identify changes to the structure and management of the drainage system, and identifies current approaches to hydrological management. It also considers land use and ecological history, both of which are intrinsically linked to, and affected by, hydrological management on the site. An important component of the review provided in this Chapter is the identification of data for the application of the various hydrological and hydro-ecological approaches that are the foundation of this thesis. For this purpose, the physical character of the wetland catchment is described, including its soils, flora and fauna. In later Chapters, some of these data are applied within water balance-based modelling approaches that are used to address some of the key management issues evident on the wetland. These are also identified in this Chapter.

2.2. History of the Pevensey Levels wetland

2.2.1 RECLAMATION

The history of the Pevensey Levels is dominated by the changing relationship between land and the sea (Dulley, 1966). The first available record describing the site dates from Roman times, at which time the area was a wide, tidally-influenced bay (all land below 4 m O.D. was submerged at high tide), studded with an archipelago of small islands or eyots (Salzmann, 1910). Many of these eyots have retained their names to the present day, reflecting their past geomorphic character (*e.g.* Horseye, Chilley). The Roman garrison fort of Anderida, dating from the 3rd Century A.D., was sited on a peninsula jutting out into the bay. A water gate found at the castle which still stands today suggests that at this time the sea came up to the castle walls (Steel, 1976) (Plate 2.1). There is however no evidence of drainage during Roman times, the first record being a mention of dykes in two Anglo-Saxon charters in 772 (Steel, 1976). These charters describe a series of dykes in the north-western part of the marsh (Barnhorn) as ‘old’ (Salzmann, 1910), implying that some attempts to reclaim the land had been made. However, the fact that these charters describe the area as salt marsh, suggests that if reclamation had been attempted, it had been unsuccessful.

(a)

(b)

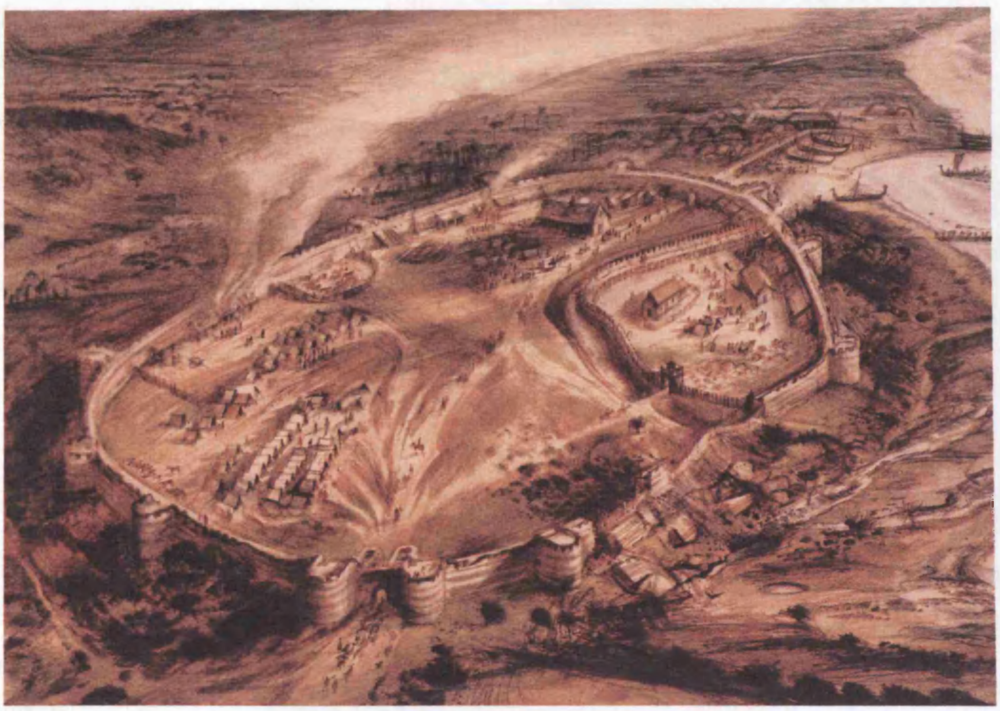


Plate 2.1. Pevensey Castle *c.* 1066 (a) and Present (b) (Reproduced from English Heritage postcards)

Pevensey holds a particularly prominent position in British history as the landing place of William the Conqueror in 1066. The first Norman church in England was also built here. Following his victory at the Battle of Hastings, William gave Pevensey to his half brother, Robert de Mortain, and the town became an important port. This coincided with the apogee of the town's prosperity. The port, although not especially large, had a regular trade of coasting vessels (Farrant, undated), providing a natural outlet for the forest products of the Weald. It was also a member of the *Cinque Ports* of Hastings. At this time, the Levels themselves were used mainly for the production of salt, a common form of exploitation in many coastal wet grasslands areas (Section 1.3.1). Figure 2.2 shows the location of saltworkings on the wetland as identified by archeological investigations in the area. Over 100 saltworks are ascribed to areas bordering the Levels in the Domesday Book of 1066, with 34 in the valley between Hooe and Barnhorn, and others on banks of the Old and Pevensey Havens (Dulley, 1966).

The centuries following the Norman conquest saw the first works aimed towards reclaiming the Pevensey Levels from the sea. Attempts to reclaim the land were only possible due to the existence of the Crumbles shingle ridge, which bound the bay at its southern end. The origin of the Crumbles shingle is in the flints eroded from the chalk cliffs of the South Downs to the west (Burrin, 1982). From about the 8^{th} Century, the eastwards drift of shingle afforded increasing protection to the bay (Table 2.1), allowing the establishment of salt marsh. In 1180, Otham Abbey was founded by Ralph de Deine, who granted parishioners his *'new marsh'*, indicating that enclosure and drainage was at that time in progress (Saltzmann, 1910). By 1200 a square ditched enclosure called Moat Marsh had been made (Dulley, 1966) (Figure 2.2) and by 1250, most of the Mountney Level had been reclaimed (Steel, 1976).

Reclamation was achieved by enclosing portions of land within sea walls or dykes, in a process known as '*inning*'. This practice became increasingly common from 1250 onwards and was possible mainly due to the large scale embanking of the major channels crossing the wetland (Figure 2.2), limiting tidal influences in saltmarsh areas. In 1289, Roger Lewkenor and Luke de la Gare were appointed as Commissioners for Sewers in Sussex, dealing with all matters pertaining to flooding and drainage on urban and agricultural land. Representing the first administrative structure devoted to the regulation of hydrology of the Pevensey Levels, they would have undoubtedly been involved in reclamation, where flood risk was an important concern.

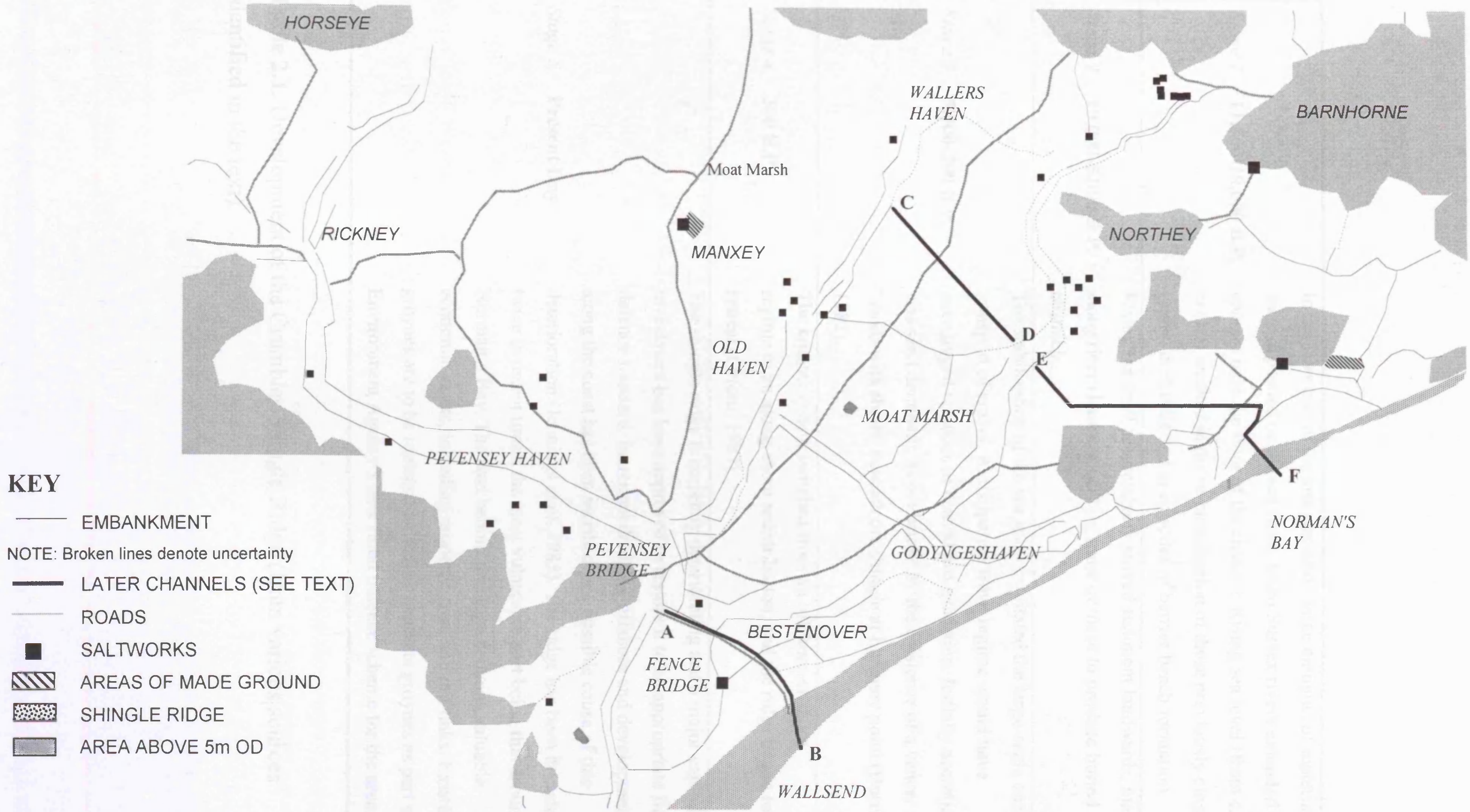


Figure 2.2. Evolution of the drainage system of the Pevensey Levels wetland 1200-1700 (from Dulley, 1966).

Stage 1	**11,000-10,000 B.P.**	In the later part of the last glaciation large amounts of materials were deposited on the sea floor, when Sussex rivers extended their courses onto the floor of the channel. Rising sea level (from c. -40 m OD) resulted in the re-mobilisation of these previously closed littoral drift cells and in episodes of barrier beach formation.
Stage 2	**10,000-5,000 B.P.**	Rising sea level progressively moved sediment landwards, mainly along river channels infilling some of these to produce buried channels.
Stage 3	**5,000-300 B.P.**	The stabilization of the sea level facilitated the large-scale onshore transport of gravel. A dissipative wave regime would have encouraged episodes of coastal pro-gradation. Initially accretion was sand dominant, as evidenced by the existence of a veneer of 'sands with shells' beneath the shingle at Langney point (Burrin, 1982).
Stage 4	**300 B.P**	The coastal system switched from an offshore-onshore regime to an along-shore redistribution, and the ridge began to retreat (Orford, 1987).
Stage 5	**Present Day**	The shingle ridge is currently deteriorating and a major capital investment has been approved to restore it to an appropriate flood defence standard. Increased dredging offshore and development along the coast has been ascribed as a possible cause of this deterioration (Jennings *et al.*,1985). The ridge has been breached twice in recent times, the most vulnerable part being that around Norman's Bay. The land behind the ridge contains valuable economic assets, including property, road and rail links. Existing groynes are to be replaced by longer concrete groynes as part of the Environment Agency's new flood defence scheme for the area.

Table 2.1. Development of the Crumbles Shingle Ridge (from various sources identified in the text).

One of the most important features of initial reclamation attempts was the large degree of interaction required between the different landowners. Following reclamation, the owners of newly enclosed lands found it necessary to guard themselves against un-neighbourly conduct that could imperil the precarious balance between land and sea (Dulley, 1966). There were however no formalised regulations until those of Romney Marsh were adopted in the 15th Century (Dulley, 1966), and it is likely that prior to that, the Commissioners for Sewers would have acted as mediators between landowners. Arrangements had constantly to be made for the drainage of one property by means of ditches running through another (Salzmann, 1910). The Abbot of Battle for example, granted part of his marshland to his neighbour, William of Northey, in return for the right to drain the rest through William's land, that lay between the Abbey and the sea (Dulley, 1966). Similarly, the Porter of Pevensey, granted the monks of Lewes the free passage of water through the marsh to the mill of Langney, apparently worked by the tides (Salzmann, 1910). The need for such agreements is illustrated by one case in particular. Between 1336 and 1342, 58 acres of land near the port of Pevensey were reclaimed without permission from the king (Dulley, 1966). Following the *inning* of these lands, the course of water to the port of Pevensey was restricted, diminishing the scouring effect of tidal water and the port started to silt up. This initiated the demise of Pevensey as an operational port, creating widespread flooding in the valley of the Wallers Haven in 1340 (Salzmann, 1910).

Problems associated with initial attempts to reclaim the marsh were addressed by major changes to the structure of the drainage system. In 1396 a new cut was made between Fence Bridge and Wallsend (A – B in Figure 2.2) in an attempt to extend the mouth southwards. The Old Haven was also cleared to increase the conveyance of water into the new cut. This however, failed to reduce the risk of flooding and, in 1402 a new cut had to be made between Dowle's Corner and Reynold's Gut (C – D in Figure 2.2). This cut represented a massive change to the organisation of the drainage system, effectively sub-dividing the Pevensey Levels into two distinct hydrological units. Prior to the construction of the cut, both the Pevensey Haven and the River Ashbourne flowed to the outfall at Fence Bridge. The new cuts however diverted the Ashbourne towards Godyngeshaven, an area of slack water draining the uplands around Barnhorn(e), Hooe and Bexhill (see Figure 2.2).

The construction of these cuts would have been a major operation. Dulley (1966) has estimated that the construction of a sewer, eight furlongs long, 10 metres wide and two metres deep would have occupied 100 men for a month. Even then there was no guarantee of success. Indeed, the new cut between Dowle's Corner and Reynolds Gut soon proved insufficient, and by1455 a new channel had to be constructed between Godyngeshaven and Normans Bay (E – F in Figure 2.2). This established the present course of the Wallers Haven and finally created two discernible hydrological units within the catchment. This feature was promoted by the construction of a sluice on the Old Haven, limiting the movement of water in the Wallers Haven towards Fence Bridge.

However, even after such large scale reorganisation, the marsh was still subject to flooding. Neglect of the sea walls in the 16th century resulted in the partial reassertion of marine influences on the marsh. Indeed, most of the seaward flank of the marsh, known as Bestenover (Figure 2.2), was reduced to salt marsh in 1594, as the eastward drift of sedimentary material generated by the degradation of the shingle ridge blocked tidal channels acting as fresh water outfalls (Dulley, 1966). This created a long narrow channel running perpendicular to the ridge as far as the original mouth at Godyngeshaven (Salzmann, 1910). The response was the construction of a new cut for the Pevensey Haven in 1630, and the erection of groynes to stop shingle drifting across the new mouth in 1634 (Dulley, 1966).

This finally allowed the *inning* of the Pevensey Levels, which was completed by 1696, a process that was greatly aided by a reduction in the flow of the River Ashbourne due to the felling of the Wealden forest for the iron industry (Steel, 1976). The final stages of reclamation saw the demise of the port of Pevensey and concluded its membership to the *Cinque Ports* of Hastings. Whilst surveying the southern coast of England to locate a site suitable for a military naval base, Edmund Dummer *et al.* (1698) wrote

'about four or five years since vessels of 50 and 60 tuns took in their loading at the (Pevensey) Bridge... but of late a shut hath been made upon the river itself very nearo beyond which no vessel may pass..tis now our opinion this Haven is irrecoverably lost, for a vessel of 14 tuns meets with great difficulty to get within the mouth of it, therefore proper for no use in the service of the navy'.

2.2.2. 1900 TO PRESENT DAY

Apart from the continuous struggle to keep the frontage of the wetland in good repair, at the beginning of the 20th century the Pevensey Levels were much as they were at the end of the 17th century (Salzmann, 1910). The Land Utilisation Survey by Briault and Henderson (1938) reports that the Levels were still entirely under permanent grass, dry enough to support beef cattle between April and October, but still prone to periodic winter flooding. Hay crops were rare since they were perceived to be detrimental to the following years crop (East Sussex County Council, 1991) and sheep were of limited importance relative to other sites nearby, such as Romney Marsh, in Kent (Morton, 1990).

By 1930 a new incentive to improve drainage and reduce flooding was provided by the Land Drainage Act (Section 1.3.2). The Act established Local Drainage Boards (LDBs), which administered and conducted flood defence works in areas with drainage problems and had powers to levy rates on inhabitants of the district. Rates from landowners were directed towards drainage improvements, and those from occupiers towards the maintenance and running costs of these works. On the Pevensey Levels, the Act funded the widening of the outfalls of the main water- courses on the wetland and the fortification of existing flood embankments.

In November 1960 the Wallers Haven banks were overtopped and much of the marsh was inundated (Plate 2.2). This event provided the necessary impetus for the implementation of a series of pumped drainage schemes on the wetland. The first of the pumped drainage schemes had already been completed at Horsebridge in 1958, but between 1969 and 1979 a further seven drainage schemes were constructed. Only one area, the Manxey South and Pevensey Bridge Levels in the central core of the wetland, remained drained by gravity. The introduction of these schemes represented a massive overhaul of the wetland drainage system. The introduction of pumped drainage created eight distinctive hydrological units within the lowland area, which made use of the main embanked channels shown in Figure 2.2. As part of these pumped drainage schemes, the Pevensey Haven and Wallers Haven were adopted as outflow channels for the pumping stations, from where water could be conveyed out to sea on the ebb tide through tidal sluice gates at Pevensey Bay and Normans Bay.

Plate 2.2. Aerial photograph showing the Barnhorn and Star Inn sub-cathments during the floods of November 1960.

(a) (b)

(c) (d)

Plate 2.3. Photographs of key land use types on the Pevensey Levels. (a) Summer grazing (cows), (b) winter grazing (sheep) and (c) arable agriculture. (d) Part of the wetland has had nature reserve status since the 1970s.

Drainage costs of about £600 per 100 hectares associated with the schemes were paid for by the landowners, but were rapidly recouped thanks to a 50 % grant available from the Ministry of Agriculture, Fisheries and Food (MAFF) (Steel, 1976).
Pumped drainage of the marsh resulted in the redundancy of *pikes*, men who made a living picking horse mushrooms, water lily blooms, sedges, watercress and rushes for making whips (Morton, 1990). Drainage allowed both the extension of the grazing season and the proliferation of arable farming in the area. Between 1968 and 1975 some 5 % of the marsh went into arable cultivation (Figure 2.3). The completion of the drainage schemes in the late 1970s, coupled with increasing grants available from the European Community's (EC, now European Union [EU]) Common Agricultural Policy (CAP), led to the extension of arable practices to close to 20 % of the total wetland area by 1990 (Figure 2.3).

During this transition to intensive arable cropping, the drainage capabilities of the wetland were further expanded by re-profiling ditches, making them wider and deeper (Palmer, 1984) and by installing under-drainage. This occurred rather later than in other wet grassland sites in the UK because the soil was generally unsuitable for under-drainage by means of mole drains or sub-soiling (Steel, 1976). Nevertheless, the area under-drained rose from 120ha in 1976 (Steel, 1976) to 500 ha in 1978 (Glading, 1986). Under-drains were installed at a depth of 0.75m (Morton, 1990), levels similar to those applied on the North Kent Marshes (Section 1.6.3) where similar soil types are evident. A further detrimental feature was the in-filling of ditches whose function was made redundant by the installation of under-drainage (Glading, 1986), a process which in the Broads resulted in a loss of 34% of ditches between 1973 and 1981 (Driscoll, 1983b). On the Pevensey Levels, some 40% of ditches in arable areas were lost in this way (Palmer, 1984), amounting to 8% of the entire original drainage network. At the same time, natural grasslands were ploughed and re-seeded with more productive grassland communities suited to drier conditions. By 1983 arable land accounted for 12 % of the wetland. At this time only 2.4% was re-seeded grassland (Morton, 1990), but by 1991 improved vegetation accounted for 44% of the wetland area (NRA, 1991; Figure 2.3).

Figure 2.3.. Land use on the Pevensey Levels wetland in (a) 1976 (from Steel, 1976), (b) 1983 (from Glading, 1986) and (c) 1991 (from National Rivers Authority, 1991).

2.3. Soils of the Pevensey Levels wetland

There has been no formal mapping of the soils of Sussex. The soils of the Pevensey Levels have been characterised by Phillips (1995) and Parish (1996) based on cores taken at a central location within the wetland. In common with wet grassland soils described in Section 1.5, the soils of the Pevensey Levels exhibit an inter-bedded sequence of clays, silts and peats, illustrating the complex sedimentological history of the wetland associated with progressive reclamation. Soils are overlain by a distinctly humose topsoil of about 0.3m depth. Below the humose layer, a four-layer sequence is apparent within the first 15m of the alluvium over bedrock. Soft brown clays to a depth of 1.5 – 2.1m below the ground surface overlie a peat layer 0.6 – 1.8 m deep. Deeper deposits include, soft grey clays, and firm ochreous clayey silt with shelly fragments (Phillipps, 1996). The peat layer, representing 'terrestrial' phases when vegetation had time to accumulate, is fragmented in nature, as confirmed by borehole transects taken across two fields by Douglas (1995). This can be attributed to erosion by river channels following deposition (Jennings and Smyth, 1987) and the irregularity of marine transgressions during the post-glacial era (Phillipps, 1995).

The stratigraphic sequence thus coincides closely with the historical account provided in Section 2.2.1. Parish (1996) has identified a series of marine clays, which were deposited between 1000 and 1500 years BP, coincident with the Roman period when the area was a tidally influenced bay. The pedological record suggests that marine processes dominated the area until at least 800 years BP (Philipps, 1995). Clayey sands are found within the succession of silts, clays and peats, and probably relate to in-filling of over incised river beds during the late glacial period (Phillipps, 1995). This is supported by cores taken by Jones (in preparation), who has found up to 2m of coarse and fine sands mixed with clay, especially close to historically important tidal channels on the wetland, such as the Old Haven.

Surface clays are alluvial gley soils of the Newchurch Association accompanied by the non-calcareous clayey Wallasea series, both typical of reclaimed lands (Cook and Williamson, 1999) (see Section 1.5). These clay soils show ochreous mottles with increasing depth (Table 2.2), indicating persistent waterlogging at depth. Newchurch and Wallasea soil associations on the Pevensey Levels are described by Jarvis and co-workers (1984) as *'seasonally waterlogged (Wetness Class III) where drainage measures are effective, but without adequate control they are waterlogged for long* *periods in winter (Wetness Class IV)'*(see Table 1.7). Mercer (1949) has estimated the moisture content of the rootzone depth to be equivalent of 185 mm of rain, of which 145mm is available water capacity (Jarvis *et al.*, 1984).

In general, limited information is available describing the physical properties of soils on the Pevensey Levels. Available information is shown in Table 2.2 and has been compiled from Jarvis *et al.* (1984). The hydraulic conductivity (K) of clay soils on the Pevensey Levels has been measured by Douglas (1995). Pump tests conducted on four dipwells in the central part of the wetland, yielded K values ranging between 0.015 and 0.172 md^{-1} (mean 0.057 md^{-1}), within the range of values commonly quoted for alluvial clays of the Newchurch/Wallasea series (see Table 1.8). Because of the low hydraulic conductivity, fluctuations in water table levels lag behind increases in ditch water level by about seven days, and when the water table is one metre below the surface, it remains virtually unaffected by surface conditions (Mercer, 1949). The low values of K also have an important influence on cropping and cultivation. Between September and April, there are very few suitable days for landwork where the land is drained by gravity alone: 46 machinery work days during an average winter and only 17 during a wet winter (Jarvis *et al.*, 1984). In contrast, during dry summers the high clay content means that these soil types tend to bake hard, exerting large capillary suctions and allowing water from the deepest pores access to the atmosphere when it is evaporated (Phillipps, 1995). They are therefore described as droughty (Jarvis *et al.*, 1984). This feature is compounded by the fact that the roots of pasture crops extend to 1.6 m below the ground surface (Mercer, 1949), encouraging higher Potential Soil Moisture Deficits than for other crops grown on the same soils (Jarvis *et al.*, 1984).

(a)

Depth (cm)	Description	*Particle Size (%)*			pH	Organic Carbon (%)	Bulk Density (g cm^{-3})	Water Capacity (% vol <2 bar)
		Sand	Silt	Clay				
0-26	Very dark greyish brown stoneless silty-clay with nunerous yellowish brown mottles.	<1	49	50	7.4	2.2	1.15	12
26-38	Light grey to grey stoneless silty clay with many coarse dark greyish brown mottles.	<1	50	49	7.5	2.4	1.20	14
38-64	Dark greyish brown stoneless silty-clay with common medium to dark brown mottles.	<1	48	51	7.8	1.1	1.35	9
64-100	Dark reddish grey stoneless silty-clay with coarse dark brown mottles.	<1	52	47	7.7	1.0	1.40	11

(b)

Depth (cm)	Description	*Particle Size (%)*			pH	Organic Carbon (%)
		Sand	Silt	Clay		
0-30	Dark greyish brown stoneless silty clay with common very fine brown mottles.	2	50	48	6.3	2.1
30-50	Grey stoneless clay with common fine brown mottles.	1	42	57	6.7	1.1
50-70	Grey stoneless silty clay with medium strong brown mottles.	0	50	50	6.5	1.1
70-110	Reddish grey silty clay with many yellowish red mottles.	0	45	55	6.9	1.1

Table 2.2. Descriptions and properties of (a) Newchurch and (b) Wallasea series soils of the Pevensey Levels wetland (from Jarvis *et al,* 1984).

2.4. The Hydrology of the Pevensey Levels wetland

2.4.1. THE CATCHMENT OF THE PEVENSEY LEVELS WETLAND

The hydrological functioning of the Pevensey Levels wetland is affected mainly by processes occurring within two distinctive, and contrasting, catchments. The Levels themselves are a flat lowland wetland area (Plate 2.4.a), which due to its reclamation from the sea are mostly below the high tide level. The lowland area is sub-divided into a series of Levels, distinct hydrological units that are not separate entities, but part of a complex system requiring careful management to maintain their agricultural and biological value. The catchment boundary on all but one flank is delimited by higher ground, with the foothills of the Weald to the North, an outcrop of Wadhurst Clay to the east and an anticlinal Tonbridge Sands ridge to the west, all rising to about 35m O.D. The boundary of the lowland catchment of the Pevensey Levels extends north to the towns of Herstmonceux and Boreham Street, to Bexhill, Lunsford's Cross and Ninfield in the east, and westwards to Hailsham and Polegate. To the southern, seaward end, the catchment is bound by the Crumbles shingle ridge, running between Eastbourne and Bexhill (Figure 2.4).

The principal source of surface water to the wetland are a series of upland catchments which converge just north of Boreham Bridge, the northern-most point of the lowland area, to form the Wallers Haven (Figure 2.5). The Wallers Haven also drains 320 ha of grazing marsh, lying between the confluence of the upland streams and Boreham Bridge. The catchment of the Wallers Haven is composed of the watersheds of the Nunnigham, Ashbourne and Hugletts streams, as well as the smaller Ninfield stream, and has a total area of 55.2 km^2 (Mercer, 1949). Flow in the three major streams have been gauged since the establishment of the Rivers Board Act of 1948, and gauges are currently the responsibility of the Environment Agency (EA) who also hold relevant data. Of the three main streams, the largest contributions are provided by the Ashbourne stream, and the smallest by the Hugletts stream (Table 2.3). These streams drain the uplands north of Herstmonceux, extending as far as Battle and Ninfield. In contrast to the Pevensey Levels, the watersheds of these tributaries are characterised by sharp relief and deep valleys (Plate 2.4.b). The catchments of the Hugletts and Ashbourne are heavily wooded, but in the catchment of the Nunnigham Stream, a large proportion of land is arable (National Rivers Archive, 1995a; 1995b; 1995c).

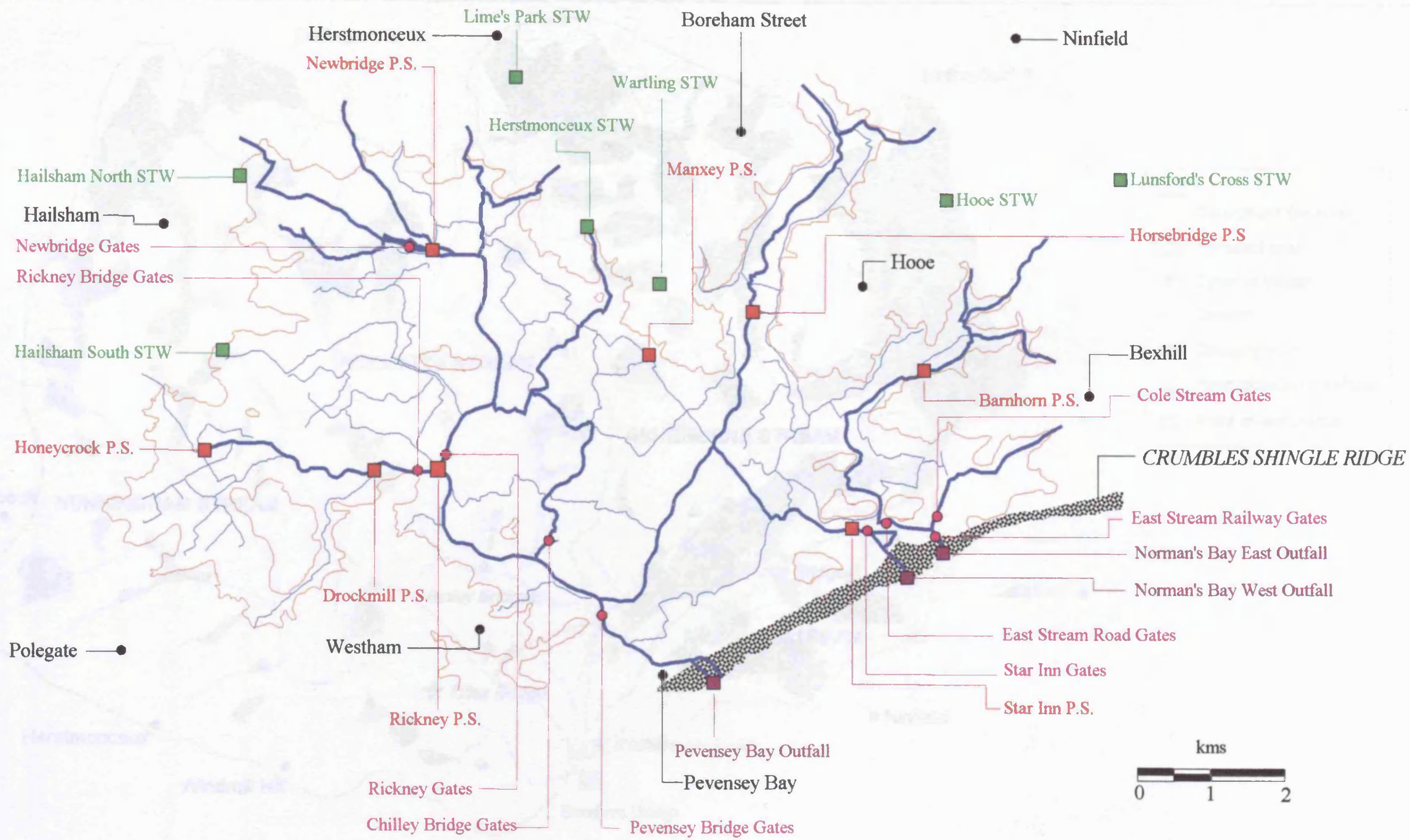


Figure 2.4.. The lowland catchment of the Pevensey Levels wetland, including embanked channels (thick blue line), Internal Drainage Board channels (thin blue line), pumping stations (P.S.), sewage treatment works (STW), gates, marine outfalls and towns.

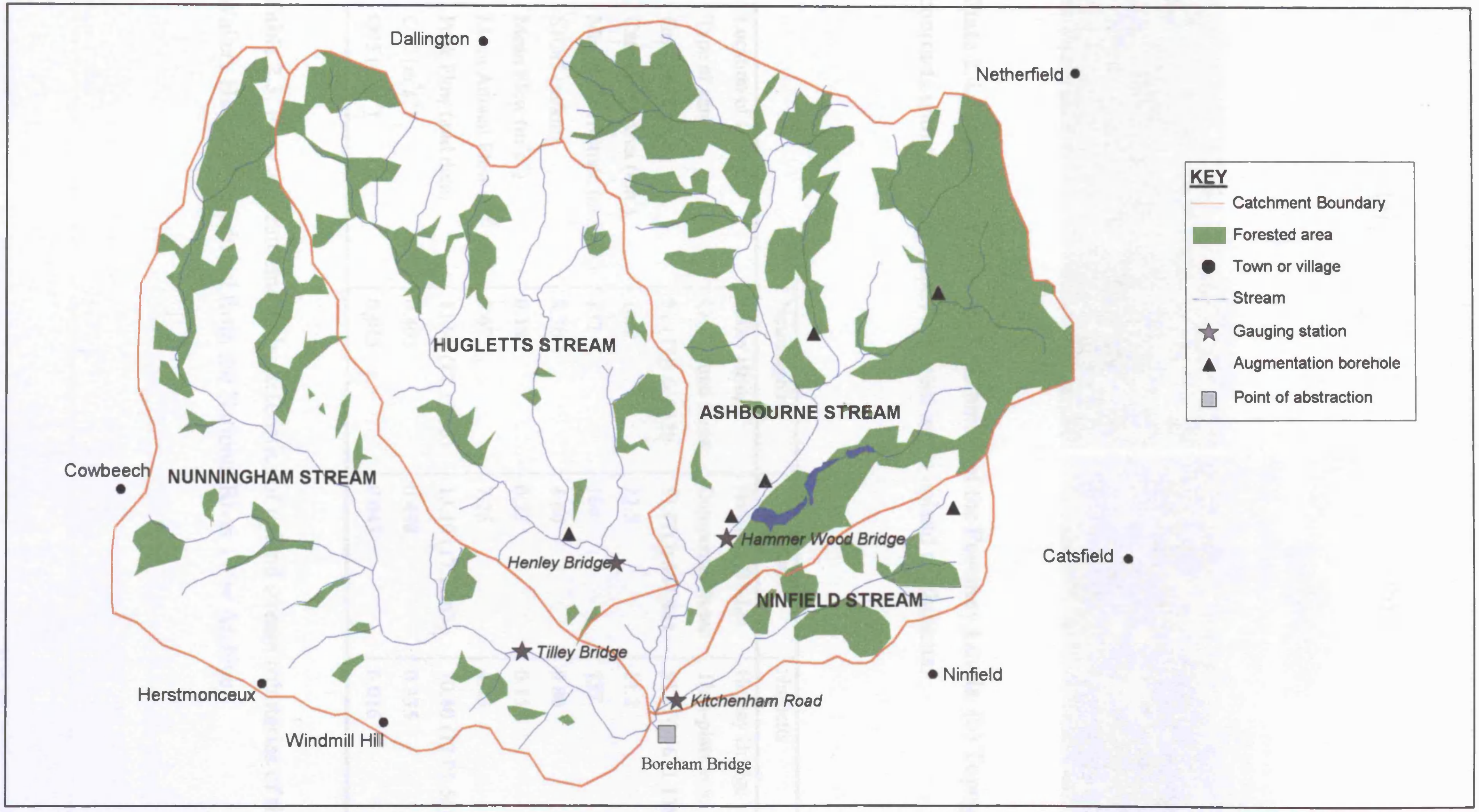


Figure 2.5. The upland catchments of the Pevensey Levels wetland, showing gauging stations and augmentation boreholes.

(a) (b)

Plate 2.4. (a)View of the lowland catchment of the Pevensey Levels. (b) Topography contrasts with the steep slopes apparent in the upland catchments.

	Nunningham	Ashbourne	Hugletts
Location of Gauge	Tilley Bridge	Hammer Bridge	Henley Bridge
Type of gauge	Compound flume	Compound flume	Thin-plate weir
Grid Reference	51 (TQ) 662129	51 (TQ) 684 441	51 (TQ) 671 138
Catchment Area (km^2)	16.9	22.5	14.2
Maximum Altitude (m O.D.)	137	169	137
S1085 (m/km)	8.30	4.60	9.80
Mean Flow (m^3s^{-1})	0.18	0.23	0.15
Mean Annual Flow (10^6 m^3)	5.67	7.25	4.73
Peak Flow (and date)	11.90 (17.11.63)	13.10 (17.11.63)	10.40 (12.01.56)
Q10 (m^3s^{-1})	0.403	0.498	0.335
Q95 (m^3s^{-1})	0.013	0.043	0.016

Table 2.3. Flow and catchment characteristics of upland stream tributaries of the Wallers Haven (Reproduced from the National River Flow Archive).

2.4.2. CLIMATE

The Pevensey Levels lie in an area of average rainfall for South-East England (Steel, 1976), receiving a mean total of 750mm of rain per year (Southern Water Authority, 1971). An eastwards decrease in rainfall is evident from analysis of long-term mean monthly rainfall (1961-1990) (Douglas, 1993) provided by eight raingauges located within the catchment of the wetland (Table 2.4). This is because prevailing winds are south westerlies and loose a considerable proportion of rainfall when passing over the chalk block immediately west of the Levels, a pattern that is evident at the regional scale (Southern Water Authority, 1973b). Comparative analyses of rainfall data describing the long term mean monthly rainfall for the periods 1941-1970 and 1961-1990 have indicated a 5% increase in recent years (Douglas, 1993).

In contrast to data availability for the estimation of areal rainfall over the Pevensey Levels, there is only one climate station within the catchment boundary providing the necessary climatic data for the estimation of evaporation and evapotranspiration. The Horseye Meteorological station is located close to the centre of the wetland, and is therefore highly representative of site conditions. Indeed, the raingauge located there has traditionally been that employed to describe rainfall over the wetland (Douglas, 1993). The station, shown in Plate 2.5.a, measures daily sunshine hours, windspeed, and wet and dry bulb temperatures (Russell Long, Environment Agency Hydrometry, Pers. Comm.). This is complemented by the presence of a Met Office type evaporation pan (Plate 2.5.b), providing estimates of open water evaporation. All instruments are read on a daily basis at 0900. Data provided by these instruments are employed to calculate evapotranspiration based on the Penman method (Loat, 1994). The above average amount of sunshine (the mean daily level between 1888-1976 was 4.96 hours per day), high mean annual temperatures (10.4°C) and the strong winds that sweep across the marsh result in high rates of evaporation (Steel, 1976). The Sussex Water Authority (1973c) suggest mean annual potential evapotranspiration (1941-1970) calculated by the Penman method is 550mm, although slightly higher values are reported for the seaward areas of the wetland.

Raingauge	Type	Altitude (m OD)	NGR East North	JAN (mm)	FEB (mm)	MAR (mm)	APR (mm)	MAY (mm)	JUN (mm)	JUL (mm)	AUG (mm)	SEP (mm)	OCT (mm)	NOV (mm)	DEC (mm)	ANNUAL (mm)
Hailsham	Storage	N/A	5588 1092	84	52	61	51	45	54	51	57	70	86	95	84	792
Eastbourne	Tipping bucket	7	5611 0980	82	54	62	50	49	49	47	55	72	91	96	83	790
Horseye	Storage	6	5627 1083	82	53	60	48	44	51	46	54	67	86	92	80	763
Flowers Green	Storage	31	5638 1115	77	50	57	48	46	53	47	57	67	86	89	78	755
Pevensey Bay	Tipping bucket	5	5662 1043	85	53	60	47	46	49	47	52	69	92	98	84	782
Hooe	Storage	30	5678 1087	74	49	55	46	44	50	43	52	64	84	88	73	722
Hazards Green	Tipping bucket	23	5682 1122	79	52	59	51	46	52	47	56	65	86	91	79	763
Barnhorn	Storage	29	5697 1078	70	48	52	46	42	46	42	48	61	79	85	71	690
Bexhill	Storage	4	5737 1072	77	50	56	47	45	48	44	49	64	82	90	77	729

Table 2.4. Mean monthly and annual rainfall 1961-1990 for raingauges located within the lowland catchment of the Pevensey Levels wetland (from Environment Agency data).

(a) (b)

Plate 2.5. (a) Generalised view of the Horseye climate station, showing raingauges, Campbell stokes sunshine recorder and Stevenson's screen and (b) the MetOffice Type evaporation pan providing estimates of open water evaporation.

(a) (b)

(c) (d)

Plate 2.6. Components of the drainage system of the Pevensey Levels wetland: (a) embanked channels, (b) large retention gates, (c) pumping stations and (d) penning board sluices on minor channels.

2.4.3. THE PUMPED CATCHMENTS

Since the 1960s and 1970s the hydrology of the lowland catchment of the Pevensey Levels wetland has been dominated by the influence of the pumps. Specific details of the dates of construction of these pumps are given in Table 2.5. The installation of these pumping stations has created a series of hydrological sub-catchments within the lowland complex, capable of operating independently of one another, although the entire lowland ditch network can be or isolated from this system if required. There are currently seven pump-drained sub-catchments (Waterlot, Barnhorn, Star Inn, Horse Eye and Down, Manxey, Whelpley and Glynleigh) and a gravity drained sub-catchment composed of the so-called Pevensey Bridge and Manxey South Levels. These sub-catchments are shown in Figure 2.6.

The hydrology of all pumped catchments is controlled by a pumping station located at the head of the catchment (Plate 2.6.c). Details of each of the pumping stations on the wetland are given in Table 2.5. The organisation of the drainage network within each pumped sub-catchment adheres closely to that shown in Figure 1.9. The drainage system is organized in such a way that farmers can, within an upper and lower limit, maintain water levels in their ditches at any desired level at any time of year (Glading, 1986). Channels leading to pumping stations are IDB channels (Table 2.6) and are generally embanked.

The pumps are operated by the IDB, which on the Pevensey Levels is the Environment Agency. The functioning of the pumps is fully automated and controlled by the water level in the pumped drain. Two electrodes are located at the pumping station, the pump on and off electrodes. The pump-on electrode is set at a higher level than the pump-off electrode (Table 2.7). When the water level has risen sufficiently to reach the pump-on electrode, the pump switches on. As pumping progresses and water is evacuated from the drain, the water level reaches the pump-off electrode (Table 2.7). This process continues until storage in the drain and any areas connected to it has stabilised below the pump-on electrode level, which is generally set to 'design' levels for agriculture (Brian Deeprose, Flood Defence, Environment Agency, Pers. Comm.). Under intense rainfall conditions, this leads to large, rapid cyclical variations in water level at the pumping station, as pumps switch on and off.

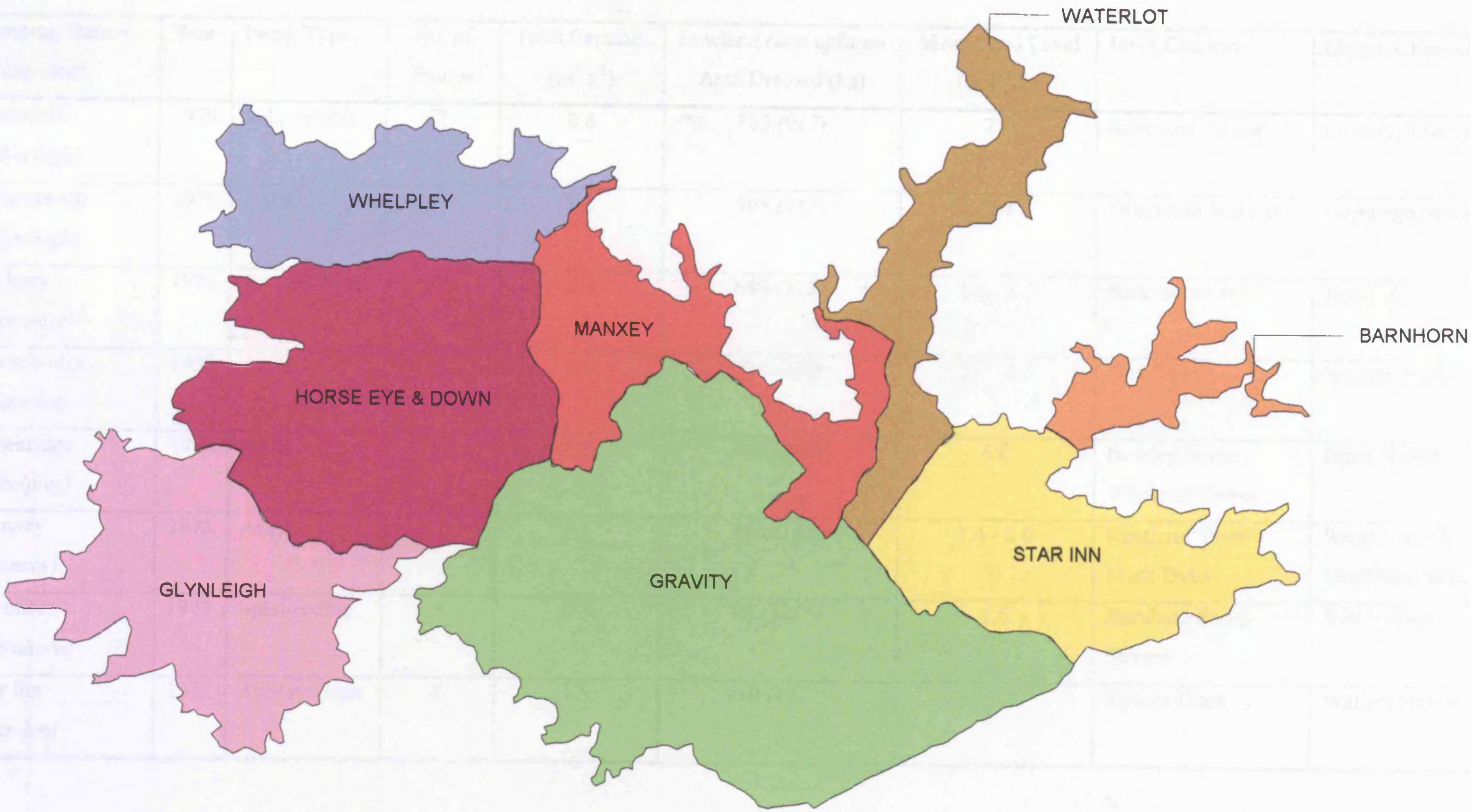


Figure 2.6. Hydrological sub-catchments of the Pevensey Levels wetland.

Pumping Station *(Pump unit)*	Year	Pump Type	No. of Pumps	Total Capacity ($m^3 s^{-1}$)	Lowland *(and upland)* Area Drained (ha)	Mean Land Level (m O.D.)	Input Channel	Output Channel
Drockmill *(Glynleigh)*	1979	Submersible	2	0.8	505 *(917)*	2.0	Saltmarsh Sewer	Glynleigh Sewer
Honeycrock *(Glynleigh)*	1979	Axial	3	3.2	505 *(917)*	2.0	Drockmill Hill Gut	Glynleigh Sewer
Rickney *(Horseeye)*	1975	Archimedean	2	2.2	680 *(222)*	1.0 - 1.5	Rickney Sewer	Hurst Haven
Horsebridge *(Waterlot)*	1959	Axial	2	1.5	320 *(610)*	2.0 - 3.5	Guy Stream	Wallers Haven
Newbridge *(Whelpley)*	1969	Axial	2	0.8	440 *(220)*	6.0	Bowley Sewer Whelpley Sewer	Hurst Haven
Manxey *(Manxey)*	1975	Axial	2	1.1	466 *(162)*	1.4 - 2.0	Kentland Sewer Mark Dyke	Burgh Fleet & Monkham Sewer
Barnhorn *(Barnhorn)*	1975	Submersible	3	0.3	84 *(1215)*	1.5	Barnhorn Ponds Stream	East Stream
Star Inn *(Star Inn)*	1975	Archimedean	2	1.3	910 *(1215)*	1.75	Stream Ditch	Wallers Haven

Table 2.5. Characteristics of pumping stations and pump-drained areas of the Pevensey Levels wetland (from Blackmore, 1993).

Type of Channel	Channel names
Embanked channels	Pevensey Haven, Wallers Haven, Glynleigh Sewer , Hurst Haven, East Stream
IDB leading to pumps	[1]Saltmarsh Sewer, Downwash Ditch, Drockmill Hill Gut, [2]Rickney Sewer, Down Sewer, [3]Whelpley Sewer, Bowley Sewer, [4]Mark Dyke, Kentland Sewer, [5]Guy Stream, Waterlot Stream, [6]Stream Ditch, [7]Barnhorn Ponds Stream
Other IDB channels	[1]Marland Sewer, Holm Sewer, Shepham Sewer, Otham Court Ditch, Duckpuddle, [2]Horse Eye Sewer, Crossing Sewer , Snapsons Sewer, Drove Sewer, Lewens Sewer. White Dyke, [3]Whelpley Sewer, Sackville Sewer, Magham Sewer, Puckeridge Stream, Iron Stream, Pevensey Mill Stream, [4]Church Farm Ditch, Kentland Fleet, Kentland Sewer, Burgh Fleet and Monkham, Sew Ditch, Dowle Stream, [5]Inn Stream, Waterhouse Stream, Pinnock Stream, New Guy Stream, Common Stream, Dodsons Ditch, [6]Chenney Stream, Cole Stream, Stream Ditch, East Stream, Foul Ditch, Old East Stream, [7]Hooe Sewer, Whydown, Picknill Green Stream, [8]Manxey Sewer, Old Haven, Callows Stream, Wrenham Stream and Bill Gut, Chilley Stream, Church Stream
Field Scale ditches	All others

Table 2.6. Classification of channels of the Pevensey Levels wetland, based on Newbold *et al.* (1989) (see Section 1.6.1) by drainage area ([1]Glynleigh, [2]Horse Eye and Down, [3] Whelpley, [4]Manxey, [5]Waterlot, [6]Star Inn, [7]Barnhorn, [8]Gravity)

Pumping Station	Pump No.	Capacity ($m^3 s^{-1}$)	Electrode Level WINTER (m OD) ON	Electrode Level WINTER (m OD) OFF	Electrode Level SUMMER (m OD) ON	Electrode Level SUMMER (m OD) OFF	Storm Override Level (m OD)
Drockmill	1	0.40	0.70	0.50	0.70	0.50	1.5
	2	0.40	1.10	0.90	1.10	0.90	
Honeycrock	1	0.40	0.10	-0.10	0.10	-0.10	0.80
	2	1.40	0.45	0.10	0.45	0.10	
	3	1.40	0.30	0.20	0.30	0.20	
Rickney	1	1.10	OFF	OFF	0.80	0.60	-0.50
	2	1.10	0.60	0.20	OFF	OFF	
Horsebridge	1	0.75	0.61	0.30	0.61	0.30	0.30
	2	0.75	0.43	0.30	0.43	0.30	*(winter)*
Newbrdige	1	0.40	0.13	0.00	0.61	0.30	-0.20
	2	0.40	0.13	0.00	0.61	0.30	*(Cut-out)*
Manxey	1	0.55	-0.30	0.00	0.30	0.00	0.05
	2	0.55	-0.45	-0.15	-0.45	-0.15	
Barnhorn	1	0.10	-0.05	na	0.05	na	-0.40
	2	0.10	-0.01	na	-0.01	na	*(Cut-out)*
	3	0.10	0.25	na	0.25	na	
Star Inn	1	0.65	0.15	-0.10	OFF	OFF	*No Data*
	2	0.65	0.80	0.60	0.80	0.60	

Table 2.7. Pump start levels for pumping stations on the Pevensey Levels wetland (from Blackmore, 1993).

There is a strongly seasonal component to the management of the pumping stations. Summer and winter electrode levels for the pump-start and pump-stop electrode levels are different (Table 2.7). This reflects the changes in the hydrological requirements of agricultural stakeholders throughout the year. The 'design' target ditch water levels are likely to closely coincide with those shown in Figure 1.11 for different types of land use. Summer electrode levels for example, are higher than those instated during the winter months (Table 2.7). Pumping in winter is employed to maintain low ditch water levels capable of providing sufficient storage capacity for rainfall during potential waterlogging and flooding events. During the summer, higher ditch water levels are preserved to act as a water distribution system providing irrigation for grass and arable crops, drinking water for livestock and enabling the separation of fields by wet fencing.

There is no set time when summer or winter settings are instated as these are generally dependant on prevailing climatic conditions. Normally however, winter settings are instated in November and changed in April (Peter Blackmore, Flood Defence Engineer, EA, Pers. Comm. 1997). In any case, a high degree of flexibility in the water management process in pumped areas is provided by the existence of a number of pumps at each pumping station (Table 2.7). Each is controlled by a different set of on and off stop electrodes (Table 2.7), so that seasonal control can also be achieved by alternating pumps, with individual pumps being used during a particular part of the year.

Economic considerations are also an important component of the management of pumping station operation. Pumping is significantly cheaper at night, costing £1.53 kWhr-1 between 0030-0730 compared to £5.00 $kWhr^{-1}$ between 0730-1900 (Marshall, 1989). Under normal conditions, pump operation is restricted by means of a time-clock to periods of 'off peak' electricity demand and pumped drains therefore have to be engineered to have sufficient storage capacity for average events. During storm conditions, the timelock is overridden when the storm override electrode is reached, an electrode which is normally set at a higher level to the winter pump-on electrode (Blackmore, 1995) (Table 2.7).

2.4.4. EMBANKED CHANNELS

All pumping stations discharge into high-level, straightened embanked channels, which generally dissect the wetland from North to South. In many cases they also form the boundary between the different pumped sub-catchments in the lowland area. Embanked channels on the Pevensey Levels are typically 16m wide at the top and 6.5m wide at the bed, 3.25m deep with bank slopes of 1:1.5 (Blackmore, 1993). The banks are a minimum of 3m wide and are typically 1.3m above surrounding field level (Blackmore, 1993). Three main hydrological systems can be identified within the network of embanked channels. These are conceptualised in Figure 2.7. Distinction between these three systems provides the basis for a conceptual model of the hydrological functioning of the wetland, since they operate independently of each other, although a series of Internal Drainage Board channels connect them (Figure 2.7).

To the East, as well as draining upland runoff from the Nunningham, Ashbourne, Hugletts and Ninfield streams, the Wallers Haven receives pumped discharge from the Star Inn and Horsebridge stations, draining the Star Inn and Waterlot pumped sub-catchments respectively (Figure 2.7). The East Stream runs parallel to the Wallers Haven and drains upland runoff from the Hooe Sewer, Whydown and Picknill Green Streams, as well as conveying discharge from the Barnhorn pumping station. This system probably represents that which initially converged at Godyngeshaven (Section 2.2, Figure 2.2).

A third system, and by far the most extensive, are a series of embanked channels to the West of the Marsh Road. Composed in the upper reaches of the Hurst Haven and Glynleigh Haven, these two channels meet at Rickney pumping station, to form the Pevensey Haven (Figure 2.7). About two kilometres downstream of Rickney pumping station, the Chilley Stream, conveying pumped drainage from the Manxey pumping station and some from the gravity drained area converges with the Pevensey Haven (Figure 2.7). Honeycrock and Drockmill pumping stations both drain the Glynleigh pumped sub-catchment, and discharge directly into the Glynleigh Haven. Pumps at Rickney and Newbridge serving the Horseye and Down and Whelpley pumped sub-catchments respectively, discharge into the Hurst Haven (Figure 2.7).

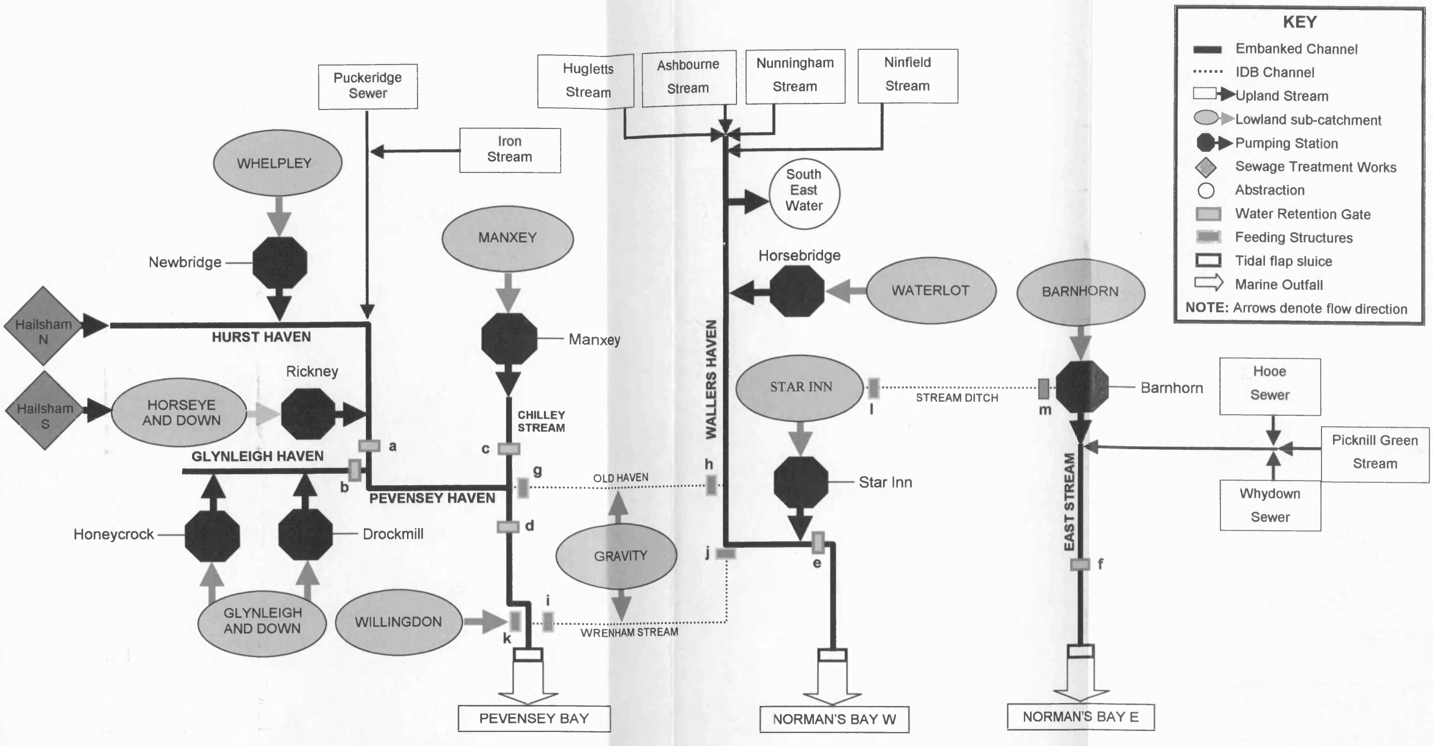


Figure 2.7. Conceptual model of the hydrological functioning of Pevensey Levels.

The supply of upland runoff to embanked channels in the west of the wetland is a fraction of that to the east (Douglas, 1993). Whilst the Wallers Haven conveys runoff from four upland catchments, there are no discernible streams feeding the western portion of the wetland. Only the Puckeridge Sewer, Iron Stream and the Hurst Haven extend significantly beyond the five metre contour line, the virtual boundary of the 'upland' area, and provide runoff to the western branch of the drainage system (Figure 2.7). In the southern branch, Honeycrock pumping station marks the source of the Glynleigh Haven and channels leading to Drockmill pumping station drain the higher land in the south-western corner of the Levels. The hydrology of the East Stream system is similar to that of the Wallers Haven, although on a smaller scale. The catchments of the Picknill Green Stream, Whydown and Hooe Sewer are all located on higher land and drain to the Barnhorn pump-drained area. However, measurements of flow during the summer of 1993 in the three largest 'minor' streams (Douglas, 1993) have shown that these provide only a fraction of the volumetric amounts conveyed by the Ashbourne, Hugletts and Nunningham streams (Table 2.8), and run dry in summer.

The distinction made between the three hydrological units within the network of embanked channels on the wetland is essentially based on the existence of marine outfalls evacuating water from the wetland out to sea (Figure 2.7): each of the channel systems identified leads to its own outfall. Because most of the land is below the high tide level, the Pevensey Levels are tide-locked for a large proportion of the day and water can only be discharged through the outfalls on the ebb tide (Blackmore, 1993). Tidal flap sluices ensure that salt water cannot access the wetland during high tides, but allows the free passage of water from embanked channels out to sea at low tide.

	Wallers Haven tributaries			East Stream tributaries		
	Hugletts	Ashbourne	Nunningham	Moorhall	Whydown	Pevensey Mill
MAY	0.050	0.097	0.007	0.005	0.012	No data
JUNE	0.030	0.069	0.017	0.004	0.007	0.005
JULY	0.021	0.058	0.022	0.001	0.006	0.003
AUG	0.017	0.071	0.016	0.001	0.003	0.002
SEPT	0.033	0.081	0.030	No data	0.006	0.003

Table 2.8. Mean daily discharge in upland tributaries of the Wallers Haven and East Stream during the summer of 1993 (from Douglas, 1993).

2.4.5. WATER LEVEL MANAGEMENT IN EMBANKED CHANNELS

The amount of water discharged from embanked channels through each of the marine outfalls can be regulated by operation of water retention gates which are typically located at their downstream end. The location of the main water level management structures on embanked channels are shown in Figures 2.4 and 2.7. Details regarding their operation are given in Table 2.9. In the case of all three embanked channel systems, a gate is located upstream of the tidal flap sluice. Downstream of these gates, the channels tend to be engineered to provide storage in times of tide lock, and may be concrete lined. Gates controlling water levels in the Wallers Haven are located at Norman's Bay (Figure 2.7). On the East Stream discharge to sea is controlled by gates under a bridge on the Bexhill road and at the Star Inn Public House. The embanked channel system to the west operates more as a cascade. Individual gates control the passage of water from the Hurst Haven, Glynleigh Haven and the Chilley Stream into the Pevensey Haven. Water levels in the Pevensey Haven are then regulated by the operation of a large gate located under Pevensey Bridge, where some structure for water control has been located since early reclamation attempts (section 2.2.1). The Pevensey Bridge Gate determines the amount of water entering the Salt Haven, the name given to the Pevensey Haven between the gate and the marine outfall. This channel can also drain the Langney Sewer, serving the Willingdon Levels around Eastbourne (Binnie and Partners, 1988), by operation of a gate at Fence Bridge.

The management of water levels in embanked channels follows the same broad principles applied for hydrological management in the pumped sub-catchments on the wetland, retaining high levels in the summer and low levels in the winter. The principles behind water level management within embanked channels of the Pevensey Levels were originally set out in Mercer (1949). Management of water levels is seasonal and primarily in response to agricultural objectives. Low winter water levels provide sufficient capacity for the storage of water pumped from the lowland, as well as upland runoff, during periods of tide lock (Douglas, 1993). Water levels in the Wallers Haven for example, must not exceed 3.00m OD, as beyond this level, bankside land is at risk from flooding (Mick Philips, Environment Agency, Sluicekeeper, Pers. Comm.).

Label	EA Ref. No.	Type	Board Max. (m OD)	Board Typical (m OD)	Pipe Invert (m OD)	U/stream watercourse	D/stream watercourse	Notes
a	R03	GATES	2.07	1.87	0.47	Hurst Haven	Pevensey Haven	Rickney automatic gates. Retain high level in Hurst Haven
b	G23	GATES	1.76	1.21	-0.77	Glynleigh Haven	Pevensey Haven	Rickney Road Bridge Gate. Retention for Glynleigh Haven
c	P07	GATES	3.26	No data	0.14	Chilley Stream	Pevensey Haven	Chilley Gates. Retention for Chilley Stream.
d	P33	GATES	No data	1.09	-0.82	Pevensey Haven	Salt Haven	Pevensey Bridge Gate
e	S36	GATES	5.61	2.91	na	Wallers Haven	Normans Bay E outfall	Star Inn Gates. Retention for Wallers Haven
f	S37	GATES	2.40	1.32	na	East Stream	Normans Bay W outfall	East Stream railway gates
g	P12	BOARD	2.54	2.43	1.36	Old Haven	Pevensey Haven	Retention for gravity area.
h	M42	BOARD	2.27	1.85	1.05	Wallers Haven	Old Haven	Feed from Wallers Haven into Gravity area via Old Haven
i	P35	GATES	3.29	2.12	0.54	Wrenham Stream	Salt Haven	Retention in Wrenham Stream for gravity area
j	P29	BOARD	2.46	2.31	na	Wallers Haven	Wrenham Stream	Feed from Wallers Haven into Gravity area via Wrenham Stream
k	No data	GATES	No data	No data	No data	Langney Sewer	Salt Haven	Fence Bridge Gates.
l	S33	BOARD	No data	No data	No data	Stream Ditch	East Stream	Feed from Stream Ditch
m	B01	VALVE	na	na	na	East Stream	Stream Ditch	Feed from East Stream (Barnhorn) into Stream Ditch (Star Inn)

Table 2.9. Inventory of water level management control structures labeled in Figure 2.6. Typical values refer to summer conditions (from Blackmore, 1993).

During the summer, higher levels are promoted. Based on the water level data provided by Mercer (1949), summer retention levels may be up to 1.2 metres higher than those in winter. Manual gravity gates located at most pumping stations on the wetland, allow water to be fed from the high level channels into the lowland if the difference in the hydraulic head across the two channels is appropriate (Blackmore, 1993). In the case of the Wallers Haven, lowland 'feeding' is not possible when water levels at Boreham Bridge recede below 1.75 m OD so that summer water levels must be carefully monitored, especially during dry spells (Mick Philipps, Environment Agency, Sluicekeeper, Pers. Comm.).

2.4.6. WATER LEVEL MANAGEMENT IN FIELD SCALE CHANNELS

In addition to 115 km of embanked and Internal Drainage Board channels, those over which the Environment Agency has direct control there are 600 km of ditches in private hands (Steel, 1976, Glading, 1986, Lindsey, 1992). Private ditches drain adjacent fields, that are frequently linked to the ditch system by a series of shallow trenches, or grips, across the field surface (Morton, 1990). The majority of field scale ditches are two to three metres wide, steep sided and below one metre in depth (Glading, 1986). Based on these dimensions, they adhere closely to the Type 1 ditches in the classification of Newbold *et al.* (1989) (Section 1.6.1.). The drainage system is organised in such a way that farmers can, within an upper and lower limit, maintain water levels in their ditches at any desired level at any time of year (Glading, 1986). Private ditches can be either linked or isolated from the main arterial watercourses by operation of sluices characteristically located at the intersection between different ditch types. There are 265 water level control structures on the Pevensey Levels, nine of which are gates located on embanked channels. Smaller sluices on private ditches are controlled manually by 'sluice keepers' in response to rainfall and the wishes of the farmers (Mick Phillips, Environment Agency, Sluicekeeper, Pers. Comm.). By far the highest proportion of all sluices on the wetland (77%) are of the penning board type and analogous in form to the structure shown in Plate 2.6.d. These structures tend to be concrete frames at the junctions between ditches, where upstream water level can be determined by the number of piles (wooden boards about 0.3m wide) inserted in the frame. A smaller percentage of all structures (15%) are valve sluices. The relative absence of this type of sluice, which can only be opened or closed, are evidence for the past and present importance of precise water level control on the wetland.

Limited information describing the hydrology of or water levels within ditches of the Pevensey Levels had been collected prior to this project. Traditionally, water levels in the ditches of the wetland were kept high, with widespread flooding common in the winter months. Boards were put in place as early as February, since marshmen always seemed to have fear of a drought (Morton, 1990). More recently, similar forms of management have been adopted on the Sussex Wildlife Trust nature reserve. Water level collected at the site clearly illustrate the different water level objectives of nature conservation and present-day agricultural stakeholders on the wetland. Water levels in the Manxey Pumped System correspond to the levels coincident with 'design' water level prescriptions for agriculture. Water levels in the Old Haven illustrate the typical role played by IDB channels. In early summer, water levels in IDB channels are maintained at levels higher than field-scale and pump drained channels to ensure their capacity for lowland feeding. In winter, water levels are generally lower than other ditches in the surrounding area to provide flood storage and drainage.

Water levels on the nature reserve, located in the gravity-drained area, are generally higher (Figure 2.8) than in the Manxey pump-drained sub-catchment. As a result, water levels in the Nature Reserve cannot be considered representative of the wetland as a whole, especially since flooding in winter and spring are encouraged for the benefit of birds and other characteristic flora and fauna of wet grasslands. However, in dry summers target water levels may not always be satisfied, as identified by Fletcher (1995), who states that '*during the summer of 1995, the summer target of 0.6m below ground surface was not attained, mainly because of evaporation during the very hot summer*'. This statement highlights the important role played by IDB ditches such as the Old Haven in maintaining water levels for the specified land use. The importance of feeding is likely to be particularly important in pump-drained areas. Ditches in pump-drained areas tend to be shallower than those in gravity drained areas and therefore retain less water (Glading, 1986) (Table 2.10). In all channels on the wetland, dredging and weed cutting is essential for the effective functioning of the drainage network (Beran, 1982b), ensuring the uninterrupted flow within ditches during drainage or feeding. On the Pevensey Levels wetland, ditches are dredged every six to seven years to remove silt and debris and restore ditch dimensions (Fletcher, 1993). The rapid colonisation of drainage ditches after clearance shows that most aquatic vegetation species on the wetland are well adapted to a periodic dredging regime (Hingley, 1979, Fletcher, 1995).

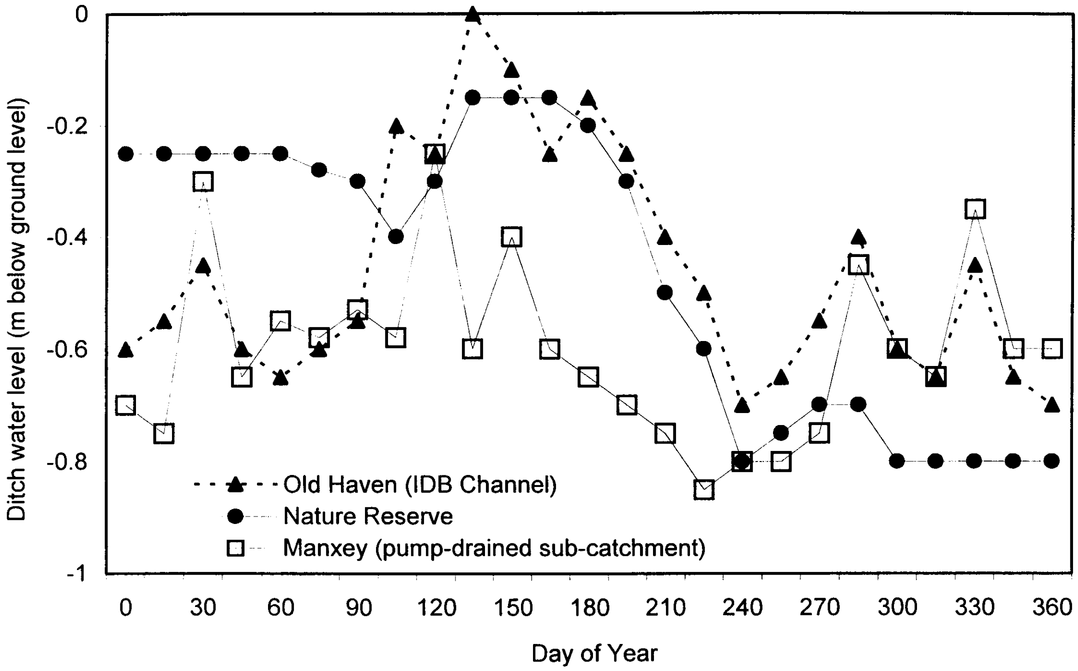


Figure 2.8. The seasonal behaviour of ditch water levels on the Sussex Wildlife Trust reserve (from Fletcher, 1993).

Water depth (m)	Pumped	Un-pumped
0 – 0.25	7.7	15.9
0.25 – 0.50	44.6	32.2
0.50 – 0.75	15.4	22.7
0.75 – 1.00	16.9	21.0
1.00+	15.4	8.2

Table 2.10. Comparison of water depths between pumped and un-pumped areas of the Pevensey Levels wetland (from Glading, 1986). Data are a percentage of all ditches surveyed.

2.4.7. ABSTRACTION

In 1947, due to the continuing failure of boreholes in the adjoining Cuckmere catchment and supplying the town of Bexhill, the Bexhill corporation approached the Catchment Board to take water for public water supply from the Wallers Haven (Mercer, 1949). Water is currently abstracted from South East Water's Hazards Green pumping station at Boreham Bridge (Figure 2.5) which is currently licensed to abstract 17,000 $m^3 day^{-1}$ (Ken Hutchinson, Environment Agency, Pers. Comm.). Abstraction from the wetland represents one of the main contentious issues associated with the management of the wetland, and has been raised by numerous wetland stakeholders. Water abstraction from both surface and ground waters has increased in tune with the population in recent years and is perceived to threaten numerous wetlands in the UK, as evidenced by English Nature's (1997) assessments of the vulnerability of 152 wetland SSSIs to abstraction, which identified the Pevensey Levels as a site where '*abstraction is an issue*' (EN, 1997). The potential impacts of abstraction on the site have been subsequently considered by South East Wateras part of the requirements of the Habitats Directive.

In particular, conflicts have arisen due to the difficulty in managing the Wallers Haven water resource to ensure sufficient water is available for both abstraction and for feeding the wetland in dry years. One of the main problems is that although the rate of flow for abstraction is fixed, the rate should ideally vary in accordance with actual river conditions (Mercer, 1949). This problem is compounded by the fact that both agricultural and public demands for water increase in drier than average years. An increasingly important issue has therefore been how to regulate abstraction. Given the turbulence of Wallers Haven flow at Boreham Bridge, it has not been possible until recently, to measure flow at the site directly (Douglas, 1993). It must be noted however, that there must at some time have been a weir or flume at Boreham Bridge, since Mercer (1949) has presented a rainfall-runoff curve for the upland catchment for the period between March 1945 and December 1946 based on data collected there. An ultrasonic river-flow gauge was installed in 1995 to address current river flow estimation problems at Boreham Bridge. The gauge however has suffered from continuous vandalism, due to its location close to a main road, and by the end of 2003 only a total of 18 months of discontinuous data had been recorded (Russell Long, Hydrometrics Section, Environment Agency, Pers. Comm.). As a result, the Environment Agency still relies on a method for the estimation of flow in the Wallers Haven to regulate abstraction, rather than direct measurement.

The method employed probably dates from the original abstraction license issued in 1947. Flow in the Wallers Haven at Boreham Bridge ($Q_{\text{Wallers Haven}}$) is calculated by addition of the recorded flows at gauging stations on the four feeder streams and application of a factor to account for the runoff generated between the gauging stations and Boreham Bridge (Ken Hutchinson, Environment Agency, Pers. Comm.), where:

$$Q_{\text{Wallers Haven}} = [Q_{\text{Nunningham}} + Q_{\text{Hugletts}} + Q_{\text{Ashbourne}} + (Q_{\text{Ninfield}} \times 1.5)] \times 1.12 \quad \text{(Equation 2.1)}$$

where $Q_{\text{Nunningham}}$ is flow in the Nunningham Stream at Tilley Bridge, Q_{Hugletts} is flow in the Hugletts Stream at Henley's Bridge, $Q_{\text{Ashbourne}}$ is flow in the Ashbourne Stream at Hammer Wood Bridge and Q_{Ninfield} is flow in the Ninfield Stream at Coombe Hill.

It has been suggested that this method over-estimates discharge in the Wallers Haven, particularly during low flows (Loat, 1994). As a result in many summers, the flow of the river will be close to, or below the Minimum Residual Flow (MRF) and therefore in breach of the abstraction licence requirements (Douglas, 1993). The MRF dates from the 1963 Water Resources Act and is the minimum discharge required *'for safeguarding public health and for meeting the requirements of agriculture, industry, water supply and the requirements of land drainage, navigation, fisheries and conservation'* (Goudie, 1991). There is however no standard way of calculating this value since it necessarily depends upon local circumstances (Petts *et al.*, 1996). The water company on the Pevensey Levels has a duty to maintain a flow equivalent to 3,410m^3/day as a condition of their licence (Douglas, 1993).

However, because of the complications apparent in measuring flow in the Wallers Haven, the minimum water level at which sluices can be used to feed the lowland area, equivalent to 1.75 m OD at Boreham Bridge, can also be employed as a surrogate MRF. In order for abstraction to continue when water levels recede beyond this level, six 'augmentation' boreholes sunk into the Ashdown Sands are operated by the water company. Two augmentation boreholes are in the catchment of Ashbourne Stream (Henley's Bridge and The Pound) and a further four are located in the catchment of the Hugletts stream (The Park, The Towerhouse, Ten Acre Gill, Burnt Barns Lane)(see Figure 2.5). All augmentation boreholes are upstream of the gauging stations on each of the rivers so that abstraction can be regulated effectively.

2.4.8. SEWAGE TREATMENT WORKS

Although surface flows in the western embanked channel system are a fraction of those in the East (Section 2.4.4), an important contribution is provided by Sewage Treatment Works serving the towns of Hailsham and Polegate. There are seven Sewage Treatment Works (STW) discharging onto the Pevensey Levels. All sewage treatment works are operated by Southern Water. However, apart from the Hailsham North and South plants, STWs at Hooe, Lunsford's Cross, Herstmonceux, Limes Park and Wartling however provide only negligible amounts of water to the wetland (Jennings, 1994). Outflow from the Hailsham South STW flows into the Horse Eye Sewer, an IDB channel in the Horseye and Down pump-drained sub-catchment, so that some effluent is pumped into embanked channels by the Rickney pumping station. Effluent from Hailsham North STW, discharges into the Harebeating stream, a tributary of the Hurst Haven.

STW discharges onto the Pevensey Levels are significant for a number of reasons. Contributions from this source represent an important element of the water balance of the Pevensey Levels hydrological system. Most of the water conveyed to the towns of Hailsham and Polegate, which the Hailsham North and South STWs serve, originates from the Arlington Reservoir located in the catchment of the river Cuckmere, to the west of the Pevensey Levels. STW inflows are therefore effectively imported from another catchment area.

Concerns have also been raised regarding the quality of this water. Research has shown that nutrient enrichment is an important processes on a number of channels in the western portion of the Levels, with phosphate concentrations at a number of monitoring points exceeding 0.4 mg l^{-1} (Jennings, 1994), the widely accepted eutrophication 'threshold' level (English Nature, 1997). Maps produced by Jennings (1994) illustrate that the channel with the greatest water quality problems are IDB channels in the Horseye and Down sub-catchment and the Hurst Haven, both of which receive inflows directly from the works. As part of their licence therefore, Southern Water are required to ensure that the quality of STW outflows is continuously monitored. Discharge from these treatment works is also monitored on a continuous basis using calibrated weirs. However, during storm conditions, much of the flow has been observed to by-pass the flumes, leading to the under-estimation of the quantity of water supplied to the wetland by the works (Loat, 1994).

2.5. Water Balance Studies on the Pevensey Levels

The water balance approach (Novitski, 1978) provides an excellent means for considering the influence of the various features of the hydrology of the Pevensey Levels as shown in Figure 2.7. There is an urgent need for water resource managers to consider the wetland water balance in making wetland management decisions (Reed, 1993). This is particularly the case in wetlands where water levels have been traditionally managed for agriculture, but where restoration seeks to raise ditch water levels with consequent increases in wetland hydrological 'demand'. Further support for the application of the approach is provided by the large number of wetland sites in the UK where abstraction for public water supply is a central theme of management related issues, but where impacts are not known in local water resource terms.

A number of water balance assessments have been conducted on the Pevensey Levels to address the water resource issues identified in Section 2.4. In particular, water balance studies on the wetland have focused on the availability of water for nature conservation-orientated management strategies, and to evaluate the sustainability of abstraction in local water resource terms. The earliest study was undertaken by Mercer (1949), who considered climatic and hydrological data for the period between March 1945 and February 1947 to establish the original abstraction licence for the Wallers Haven (Section 2.4.7). Later studies by Douglas (1993), using data for the summer of 1993, and Loat (1994), using data covering the period between October 1990 and December 1994, have also considered the issue of abstraction, although they have been primarily concerned with the sustainability of abstraction in the context of agricultural and nature conservation water level requirement. In both studies, rainfall and evaporation data have been obtained from the Horseye climate station, and employed to calculate the influence of these processes within the area covered by the SSSI.

These studies have shown the contrast between the large available winter resource and the considerable summer deficit (Figure 2.9). In particular, the important influence of the balance between rainfall and evapotranspiration has been noted. Losses by evapotranspiration between May and August are often more than two times greater than rainfall (Figure 2.10), leading to water resource deficits in late summer. In drought years this deficiency can extend well into the autumn months (Coles, 1994). Loat (1994) has shown that there was 76 % more water available on the Levels during 1994 than

1991, a drought year, whilst the period of water deficiency in 1990 lasted four months. In 1993 only two months were water deficient (Loat, 1994; Figure 2.9).

Previous studies have also found the influence of abstraction on water availability to be large. Douglas (1993) has estimated that water abstracted from the Wallers Haven can represent up to 10 % of the total flow, and that once abstraction has been considered, inputs provided by the Wallers Haven are approximately equivalent to those provided by minor upland streams (Figure 2.11). The spatial variability in water availability across the wetland has also been noted. This is mainly related to the influence of STWs, which on an annual basis, provide considerable inflows to the wetland catchment (Figure 2.12). All STWs apart from those at Hooe and Lunsford Cross, discharge onto the western part of the wetland, and as a result STW contributions to the summer water balance are 22 times larger in the west than in the east, and may provide up to four times more water than the Wallers Haven in dry summers (Douglas, 1993).

Nevertheless, comparison between the features of the hydrology of the Pevensey Levels considered in previous water balance studies, and the hydrological features of the wetland reviewed in Section 2.4 show that at least three features of the local hydrology remain un-researched. Probably of greatest importance is the lack of data concerning losses to sea. Based on the design of the drainage system of the Pevensey Levels (see Figure 2.7), the amount of water lost to sea is likely to be heavily influenced by the functioning of the pumps. However, the functioning of pump-drainage units has not been previously considered. Similarly, little is known regarding volumetric storage within the drainage network, or the importance of soil storage. Mercer (1949) provides the only information available: summer storage in the Wallers Haven is equivalent to 31.8×10^6 m^3, and 820.6×10^6 m^3 for the lowland channel network, although it is unclear whether these figures refer to the entire lowland area, or simply ditches within the sphere of influence of the Wallers Haven. Soils on the Pevensey Levels wetland contain 185mm of water, of which 145mm is available water capacity (Mercer, 1949).

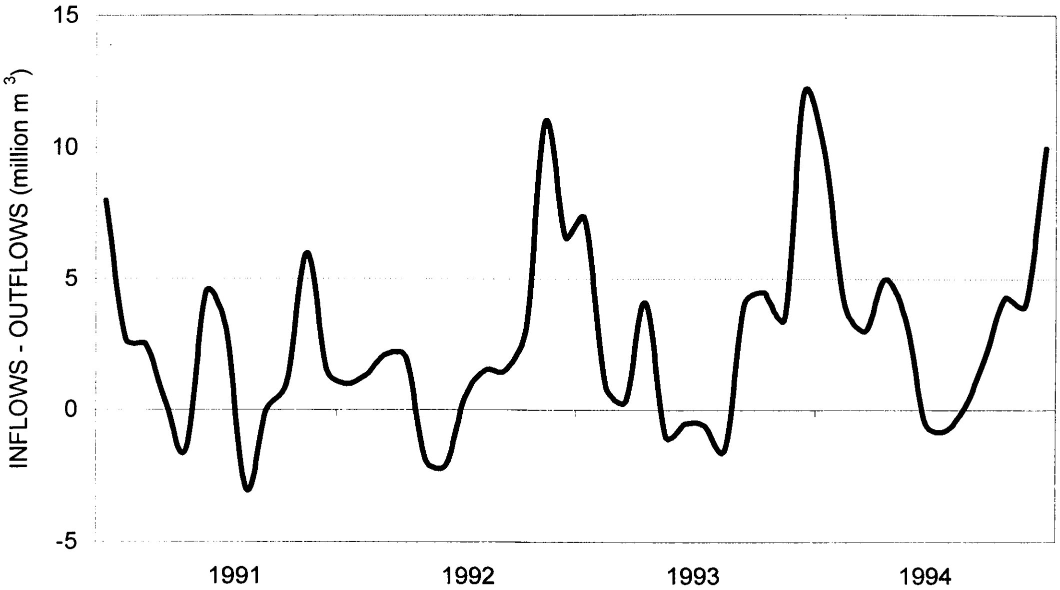


Figure 2.9. Monthly water availability 1991-94, Pevensey Levels. Data shown are the balance between inflows and outflows (excluding losses to sea). Changes in storage were not considered (from Loat, 1994).

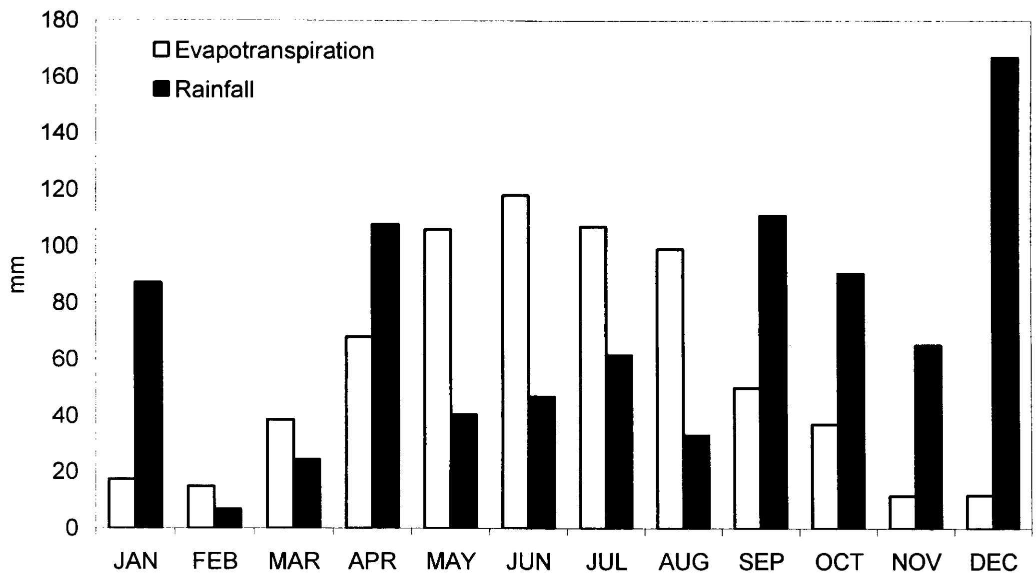


Figure 2.10. The balance between rainfall and evaporation on the Pevensey Levels, 1993 (from Douglas, 1993).

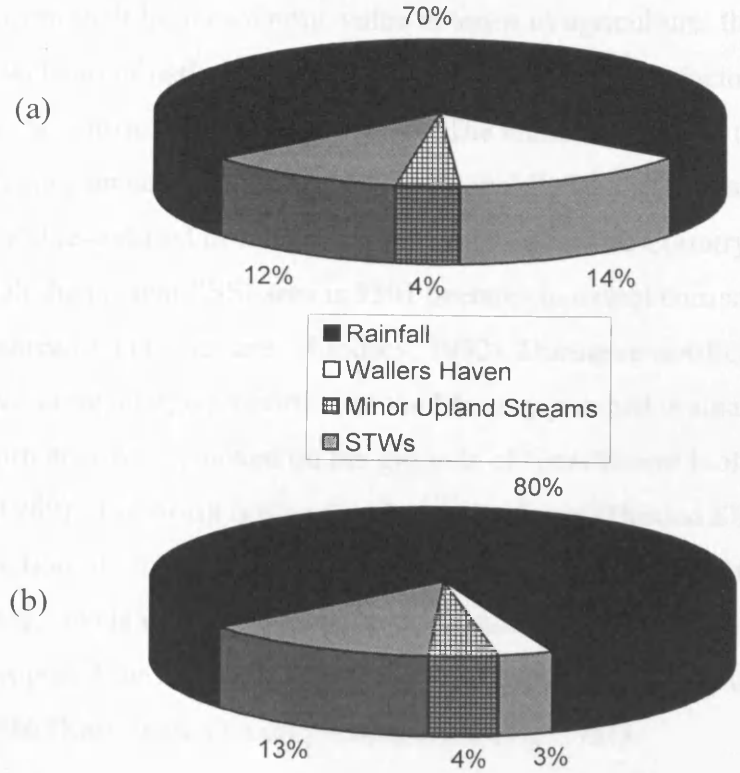


Figure 2.11. Wallers Haven flow as a percentage of total inflows in June 1993 (a) gauged flow before abstraction ('naturalised flow') and (b) after abstraction. (from Douglas, 1993).

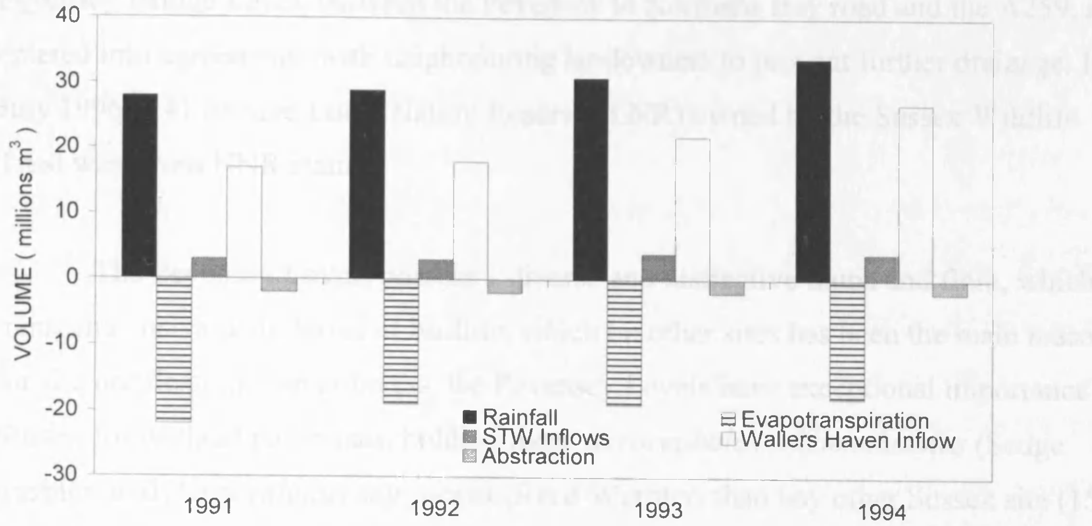


Figure 2.12. Annual water balance for the Pevensey Levels 1991-1994 (from Loat, 1994).

2.6. Biological Interest of the Pevensey Levels

Apart from their high economic value in terms of agriculture, the Pevensey Levels are also a wetland of national value in terms of biodiversity, a factor reflected in its Site of Special Scientific Interest (SSSI) status. The entire area below the five metre contour line was designated an SSSI in 1977 (National Parks and Access to Countryside Act, 1949) and re-notified in 1990 (Section 28, Wildlife and Countryside Act,1981), although the present SSSI area is 3501 hectares in extent compared to the original (1977) area of 4112 hectares (Lindsey, 1992). During re-notification, some peripheral land including a large proportion of the Manxey pumped drainage scheme and the entire Barnhorn area was removed on the grounds of 'insufficient biological interest' (Keymer *et al.*, 1989). It is worth noting that the wetland was afforded SSSI status after the introduction of all but two of the pumped drainage schemes on the wetland. The Pevensey Levels were only given Ramsar status in 1998, and in the past the site has also been proposed for Environmentally Sensitive Areas (ESA) status under the Agriculture Act, 1986 (East Sussex County Council [ESCC], 1991).

Two National Nature Reserves (NNRs) are located within the wetland complex. Both are located in the central gravity-drained core of the Levels, an area which has remained virtually unimproved since reclamation. In 1985 the Nature Conservancy Council (NCC, now English Nature) purchased 52 ha of un-drained fields on the Pevensey Bridge Level, between the Pevensey to Normans Bay road and the A259, and entered into agreements with neighbouring landowners to prevent further drainage. In July 1996, a 41 hectare Local Nature Reserve (LNR) owned by the Sussex Wildlife Trust was given NNR status.

The Pevensey Levels possess a diverse and distinctive fauna and flora, which transcends its value in terms of birdlife, which on other sites has been the main reason for site notification. Nevertheless, the Pevensey Levels have exceptional importance in Sussex for wetland passerines, holding more *Acrocephalus schoenobaenus* (Sedge warbler) and *Acrocephalus scirpaceus* (Reed Warbler) than any other Sussex site (15% of the Sussex total) as well as 20% of the Sussex population of *Motacilla flava flavissima* (Yellow wagtail) (Keymer *et al.*, 1989). In the past, the wetland has also been of national importance for the number of over-wintering *Vanellus vanellus* (Lapwing), with numbers regularly exceeding 1% of the British population, and possibly *Gallinago gallinago* (Snipe), although there has been considerable debate over the national numbers of this species. Numbers of *Pluvialis apricaria* (Golden plover) also reached levels of national importance in four years between 1972 and 1987 (Keymer *et al.*, 1989). Indeed, until recently the wetland was a candidate Special Protection Area (SPA) under the EC Birds Directive (Appendix 2.1), although it has since been removed from that list because it no longer satisfies those criteria (Basil Lindsey, English Nature, Pers. Comm.).

In terms of local flora, few rare species occur on the fields themselves (Neil Fletcher, SWT Reserve Warden, Pers. Comm.), although unimproved areas are more diverse than improved swards (Steel, 1976). Unimproved grassland areas of the wetland are dominated by *Agrostis stolonifera*, *Poa trivialis* and *Lolium perenne* with *Juncus effusus* and *Juncus inflexus* in wetter areas. On improved fields *Poa trivialis*, *Lolium perenne*, *Cynosaurus cristatus* and *Trifolium Spp.* are the main species (Glading, 1986). However, on the basis of structure, diversity and rarity of the flora, ditches on the Pevensey Levels are as rich as any ditch system in Britain (Killeen, 1994), and more so than ditches on the Broads, the Gwent Levels or the North Kent marshes (Glading, 1986). Indeed, on the Pevensey Levels it is the ditch system which provides the site with its particular ecological signature, and both the SSSI and Ramsar citations for the Pevensey Levels identify the ditch habitat as the reason for notification (Appendix 2.2 and 2.3).

There is a well developed hydroseral sequence composed of emergent, submergent and floating species, and up to 40 species have been recorded per field length in the gravity area, although in pumped ditches this value is more commonly around 25 (Glading, 1986). 177 open water or emergent taxa have been identified in surveys of the Pevensey Levels ditch (Glading, 1986). 110 of the 160 British 'aquatic' plants are represented on the site, with 37 in the rare or local categories (Whitbread and Curson, 1992), including the nationally rare *Potomageton acutifolius* and the nationally scarce *Potomageton trichoides*, *Stratiotes aloides*, *Wolffia arrhiza, Hottonia pallustris, Sium latifolium* and *Ceratophyllum submersum*. The distribution of a number of these species in the UK is shown in Figure 2.13. Dominant floating species within open water stretches are duckweed, *Lemna* spp. and frog's bit, *Hydrocharis morsus-ranae*. Submerged plants are most commonly *Potomageton spp.* or *Elodea canadiensis*. Emergent vegetation is characterised by species such as *Sparganium erectum*, *Phragmites australis,* and grasses of the family *Alimantacae* (Steel, 1976). Glading

(1986) has identified eight distinct floral assemblages in ditches on the Pevensey Levels wetland, including a distinct main channel community. Species associated with each of these floral assemblages are shown in Table 2.11. The specific botanical composition of ditches on the wetland is also related to the cycle of clearance and dredging previously highlighted as an important component of the management of ditches on the Pevensey Levels (Section 2.4.6). Bare substrate colonisers include rare species such as flowering rush (*Butomus umbellatus*), narrow-leaved water plantain (*Alisma lanceolatum*) and unbranched bur-reed (*Sparganium emersum*) as emergents, sharp leaved pondweed (*Potomageton acutifolius*) and horned pondweed (*Zannichelia palustris*) as submergents, and Canadian pondweed (*Elodea Canadiensis*) dominant in the deepest parts of the ditches (Fletcher, 1993b)

Apart from their floral interest, ditches on the Pevensey Levels are also the most important site in England for freshwater molluscs, extremely good indicators of clean, still, calcareous water (Killeen, 1994). Nineteen species of mollusc occur (Steel, 1976), including four nationally rare species (*Valvata macrostoma, Segmentina nitida, Pisidium pseudosphaerium* and *Anisus vorticulus).* 91 species of aquatic beetle, including four Red Data Book (RDB) species have also been recorded on the site (Carr, 1983) and two species, *Bidessus unistratus, Laccophillus varigatus*, occur nowhere else in England (East Sussex County Council, 1991). 193 species of diptera (Keymer *et al.*, 1989), 21 species of dragonfly, including the nationally scarce *Brachytron pratense* and *Coengrion pulchellum*, and 120 species of insect, including 21 Red Data Book species (Belton, 1987), have also been recorded.

Probably most importantly in conservation terms, is the presence of the Fen Raft Spider (*Dolomedes plantarius). Dolomedes plantarius* is the largest aquatic spider in the UK, and is only present at one other site in the UK, Redgrave and Lopham Fen, East Anglia. The Pevensey Levels represent the only expanding population in the UK, where it is closely associated with specific aquatic vegetation communities. On the Pevensey Levels, *Dolomedes* populations are largest in the gravity drained area, although they are also present in pumped-drained areas (Figure 2.14). The spider favours sunny ditches with water levels that do not normally overtop the banks (the species is poorly represented in ditches that flood regularly) and where there is a strong presence of emergent plants, such as *Stratiotes aloides*, which is a favoured plant for constructing nursery webs (Jones, 1992).

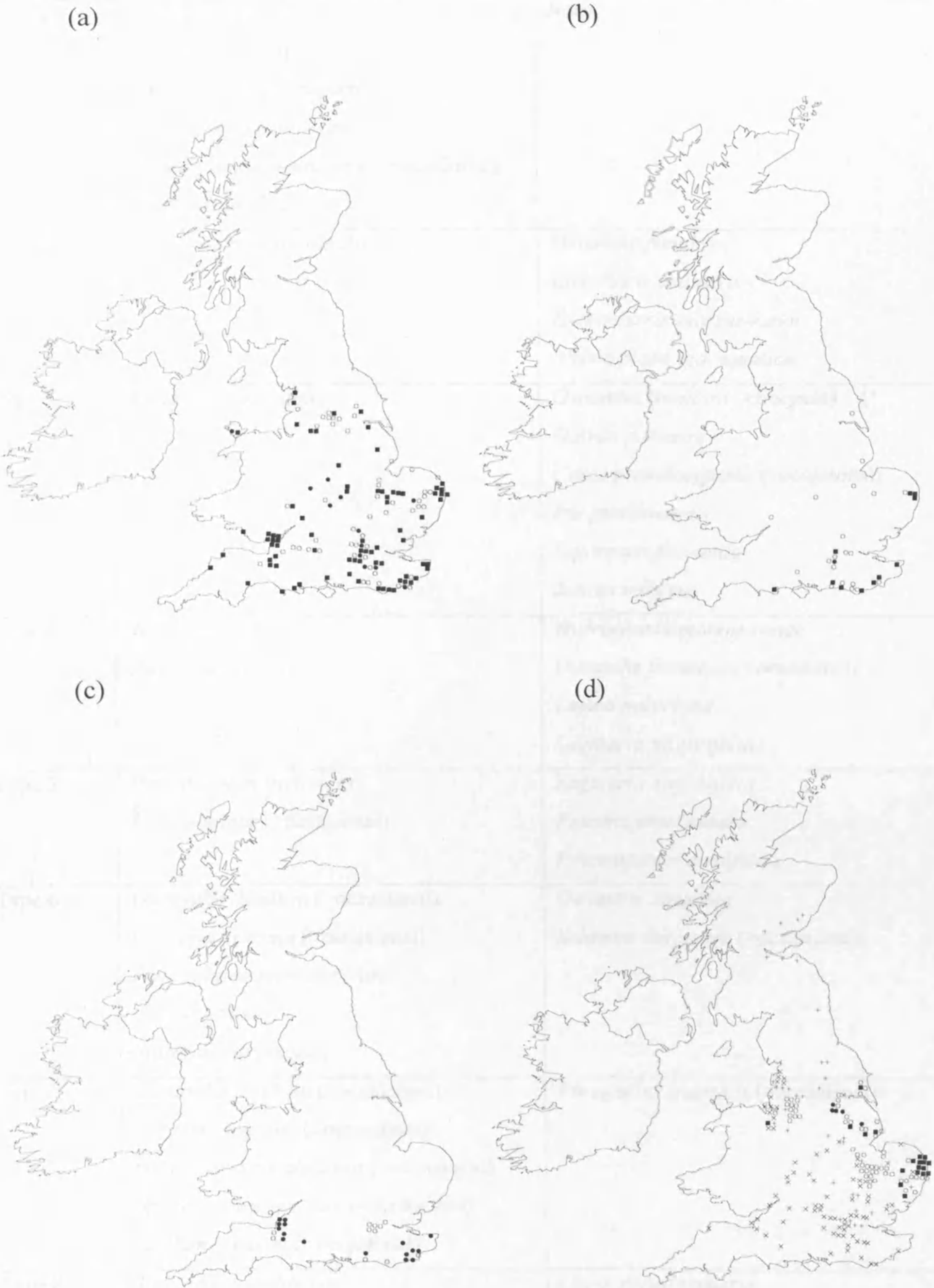


Figure 2.13. Distribution of (a) *Potamageton trichoides*, (b*)* *Potomageton acutifolius*, (c) *Wolffia Arrhiza* and (d) *Stratiotes aloides* in England (from Preston and Croft, 1997).

	Negative indicator species	Positive indicator species
Type 1	*Elodea canadensis* *Potomageton natans* *Lemna trisulca* (>frequent) *Potomageton acutifolius* *Alisma plantago-aquatica* (>occasional) *Filamentous algae*	*None*
Type 2	*Enteromorpha intestinalis*	*Oenanthe fistulosa* *Eleocharis palustris* *Hydrocharis morsus-ranae* *Alisma plantago-aquatica*
Type 3	*Sagittaria sagittifolia*	*Oenanthe fistulosa* (>frequent) *Galium palustre* *Carex pseudocyperus* (>occasional) *Iris pseudocorus* *Equisetum fluviatile* *Juncus influxus*
Type 4	*Juncus inflexus* *Juncus articulatus*	*Hydrocharis morsus-ranae* *Oenanthe fistulosa* (>occasional) *Lemna polyrhiza* *Sagittaria sagittifolia*
Type 5	*Potomageton trichoides* *Lemna minor* (>occasional)	*Sagittaria sagittifolia* *Potomageton natans* *Potomageton acutifolius*
Type 6	*Oenanthe fistulosa* (>occasional) *Glyceria maxima* (>occasional) *Hydrocharis morsus-ranae* *Juncus effusus* *Sparganium erectum*	*Oenanthe aquatica* *Solanum dulcmara* (>occasional)
Type 7	*Oenanthe fistulosa* (>occasional) *Glyceria maxima* (>occasional) *Glyceria fluitans/plicata* (>occasional) *Sparaganium erectum* (>occasional) *Galium palustre* (>occasional)	*Phragmites australis* (>occasional)
Type 8	*Phalaris arundinacea*	*Carex pseudocyperus* *Phragmites australis* (>occasional) *Juncus effusus* *Iris pseudocorus*

Table 2.11. TWINSPAN classification of aquatic flora communities on the Pevensey Levels (from Glading, 1986).

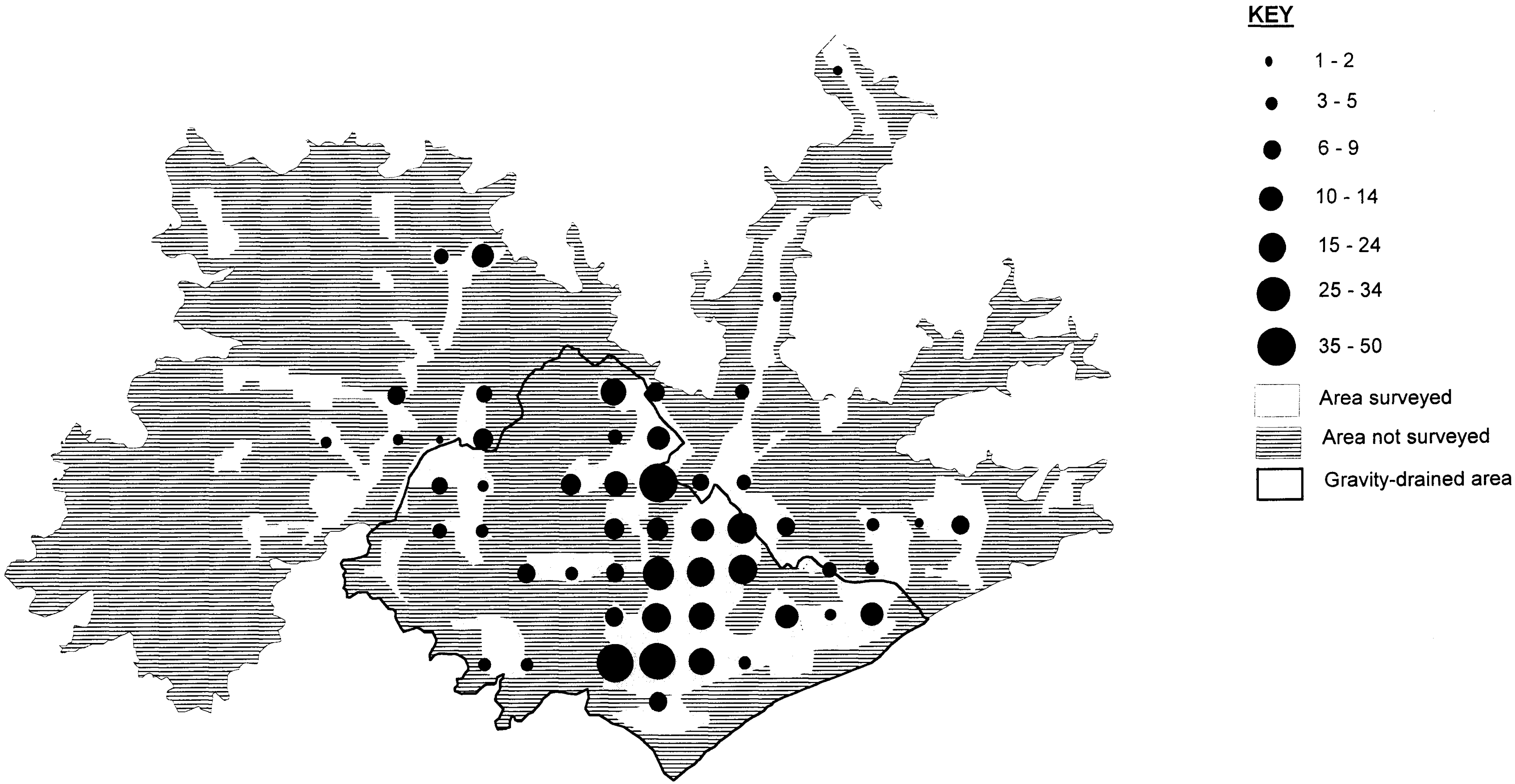


Figure 2.14. Distribution of the fen raft spider (*Dolomedes plantarius*) individuals seen totalled in 500 metre squares (from Jones, 1992).

2.7. Ecological Decline

It is widely acknowledged that although still nationally important in biodiversity terms, the ecological value of the Pevensey Levels has been in decline for a number of years (Douglas and Hart, 1993). Ecological decline mirrors the trend in Sussex as a whole, where between 1960 and 1980 10,500 ha of grazing marsh have been drained, equivalent to 66% of the original stock (Whitbread and Curson, 1992). Ecological monitoring across the Levels since the 1970s have suggested that the implementation of pumped drainage schemes, coupled with associated changes in land use patterns, have had a significant effect on the quality of the wetland habitat and the numbers and diversity of species supported by it. Similar trends are apparent in other wet grassland areas in the UK where similar drainage measures have been instated (Section 1.2.3). On the Pevensey Levels, the longest available biological records allowing the decline of the site to be quantified correspond to numbers of over-wintering birds. For other species, there is generally a dearth of baseline ecological information. Annual bird counts on the Levels have been undertaken by the Sussex Trust for Ornithology since the early 1970s. Less regular counts have been undertaken since at least 1938 (Hitchings, 1987). Counts for typical bird species of wet grassland such as Lapwing, (*Vanellus vanellus*), Snipe (*Gallinago gallinago*) and Golden plover (*Pluvialis apricaria*) between 1970 and 1994 are illustrated graphically in Figure 2.15 and shown in Table 2.12.

Throughout the 1970s numbers of Lapwing and Snipe regularly exceeded numbers of national importance (Table 2.12), and in the case of Lapwing, international importance. In the last 30 years however, the numbers of breeding and over-wintering individuals of these species has decreased. Lapwing are more sensitive to high stocking rates than site dampness (Hitchings, 1987) and their decline is evidence of the effects of the intensification of agricultural practices on the wetland that drainage has afforded. In the case of lapwing, an average 12,050 individuals were observed on the wetland between 1972 and 1982 compared with 9,000 between 1983 and 1993, a decrease of 70%. Based on the numbers of golden plover presented in studies by Hitchings (1987), Rowland and Burgess (1993) and Burgess (1994), the number of individuals of this species over-wintering on the wetland has fallen by 55% relative to numbers between 1972 and 1982. According to available data the number of snipe on the wetland has actually increased by 40% between 1972 and 1992, although this may be related to inter-annual variability or more detailed surveys in the 1990s rather than actual trends.

Year	Golden Plover	Snipe	Lapwing	Source of data
1972	800	2,000	15,300	Hitchings (1987)
1973	750	400	5,000	Hitchings (1987)
1974	850	650	10,000	Hitchings (1987)
1975	850	350	12,000	Hitchings (1987)
1976	5,300	5,000	15,000	Hitchings (1987)
1977	2,100	2,300	8,500	Hitchings (1987)
1978	No data	No data	No data	Hitchings (1987)
1979	2,700	580	26,000	Hitchings (1987)
1980	1,000	1,260	2,700	Hitchings (1987)
1981	1,200	1,000	6,000	Hitchings (1987)
1982	1,200	750	20,000	Hitchings (1987)
1983	1,025	600	8000	Hitchings (1987)
1984	2,500	1,500	16,000	Hitchings (1987)
1985	200	300	6,000	Hitchings (1987)
1986	452	361	4,000	Hitchings (1987)
1987	500	3,000	400	Hitchings (1987)
1988	200	2,000	104	Rowland and Burgess (1993)
1989	1,200	3,235	No data	Rowland and Burgess (1993)
1990	1,100	4,000	117	Rowland and Burgess (1993)
1991	260	1,500	25	Rowland and Burgess (1993)
1992	424	5,191	360	Rowland and Burgess (1993)
1993	437	360	5,200	Burgess (1994)
1994	300	513	2,283	Burgess (1994)
1% Nat. Popn.	2,500	Unknown	20,000	Burgess (1994)

Table 2.12. Maximum recorded winter numbers of snipe, lapwing and golden plover on the Pevensey Levels 1972-1994.

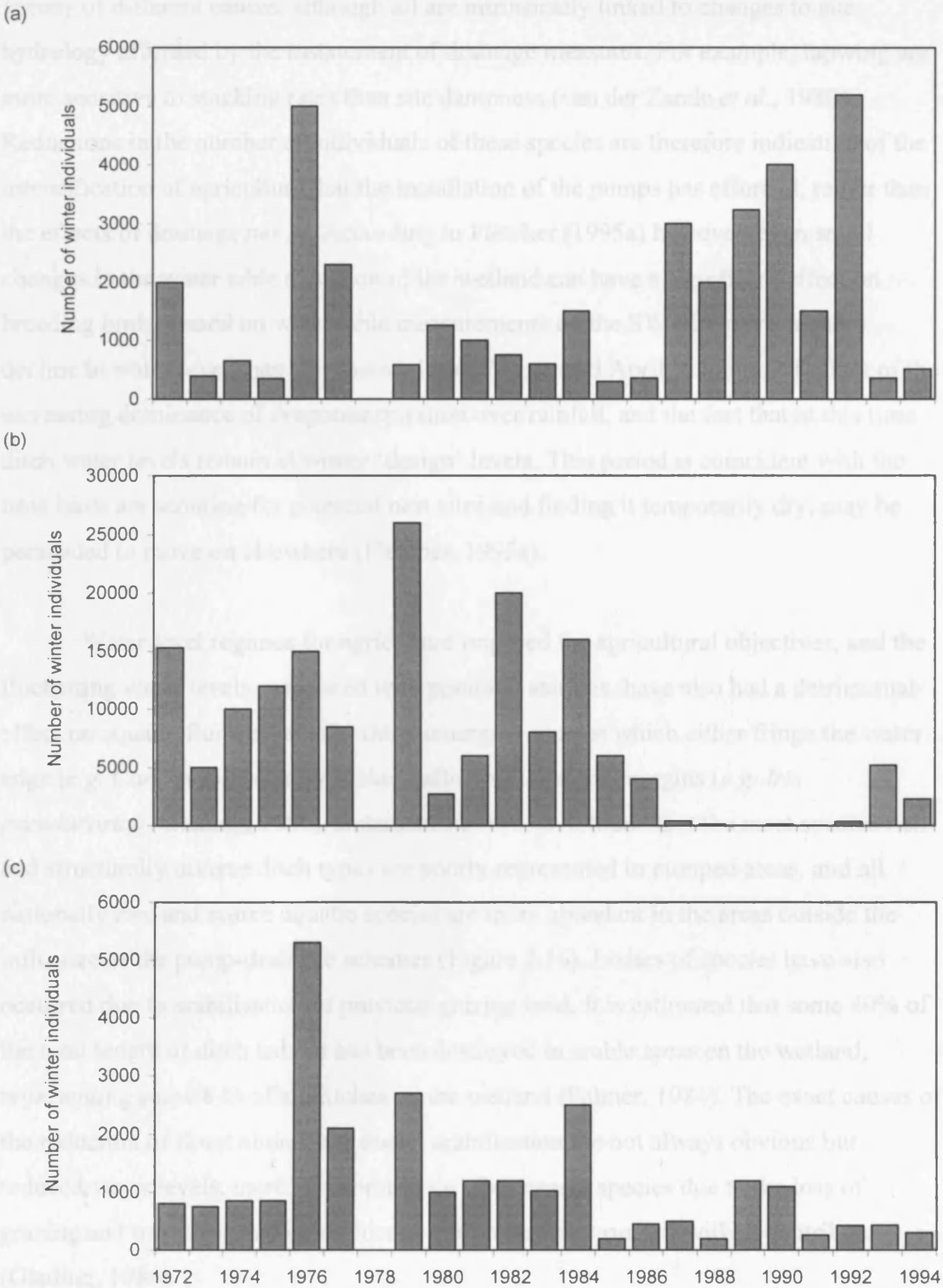


Figure 2.15. Maximum numbers of (a) snipe, (b) lapwing and (c) golden plover on the Pevensey Levels 1972-1994 (from Hitchings, 1987, Rowland and Burgess, 1993 and Burgess, 1994).

Reductions in the number of lapwing and golden plover can be ascribed to a variety of different causes, although all are intrinsically linked to changes to site hydrology afforded by the instatement of drainage measures. For example, lapwing are more sensitive to stocking rates than site dampness (van der Zande *et al.*, 1980). Reductions in the number of individuals of these species are therefore indicative of the intensification of agriculture that the installation of the pumps has afforded, rather than the effects of drainage *per se*. According to Fletcher (1995a) however, even small changes in the water table elevation of the wetland can have a significant effect on breeding birds. Based on water table measurements on the SWT Reserve, a sharp decline in water levels has been noted during March and April, a combined effect of the increasing dominance of evapotranspiration over rainfall, and the fact that at this time ditch water levels remain at winter 'design' levels. This period is coincident with the time birds are scouting for potential nest sites and finding it temporarily dry, may be persuaded to move on elsewhere (Fletcher, 1995a).

Water level regimes for agriculture imposed for agricultural objectives, and the fluctuating water levels associated with pumping stations, have also had a detrimental effect on aquatic flora, especially those emergent species which either fringe the water edge (*e.g. Carex* spp.) or grow in the shallow water at the margins (*e.g. Iris pseudacorus*) (Glading, 1986). Botanical surveys have found that the most species rich and structurally diverse ditch types are poorly represented in pumped areas, and all nationally rare and scarce aquatic species are more abundant in the areas outside the influence of the pump-drainage schemes (Figure 2.16). Losses of species have also occurred due to arabilisation of previous grazing land. It is estimated that some 40% of the total length of ditch habitat has been destroyed in arable areas on the wetland, representing some 8 % of all ditches on the wetland (Palmer, 1984). The exact causes of the reduction of floral abundance due to arabilisation are not always obvious but reduced water levels, increased dominance of emergent species due to the loss of grazing and trampling and eutrophication from fertiliser run-off will all contribute (Glading, 1986).

Differences in the flora between arable and pasture areas are generally related to vegetation structure and species composition, rather than species richness. Arable ditches tend to have a higher cover of submerged species and a lower cover of floating species than pasture ditches (Palmer, 1984). The increasing dominance of emergents may be related to changes in the water level regime of the ditches: emergent weeds grow in any water less than 1m deep and the depth under pumped schemes is generally less than 0.5m (Palmer, 1984). Especially problematic for the Internal Drainage Board (the Environment Agency) is the proliferation of floating pennywort (*Hydrocotyledon rananculoides*) on the Pevensey Levels. The plant is not a native of the British Isles, grows extremely rapidly and can dominate the channel at the peak of the growing season (Pevensey Levels Study Group, 1998; Plate 2.7.b). This has caused problems of flood defence, blocking sluices and weed screens located at pumping stations, and can out-compete floating and submerged floral species of nature conservation importance, by limiting the amount of light entering the water column.

In recent times, changes in the nutrient status of ditch water on the wetland have caused concern. Palmer (1984) for example, has noted considerably higher conductivity values in ditches surrounding arable land on the wetland than in channels surrounded by pasture, an indication of the presence of higher concentrations of fertiliser in ditch water. The proliferation of *Lemna gibba* and *Azolla filiculoides* (Plate 2.7.a) in particular, has raised concerns over the effect of spraying arable fields with sewage sludge, a practice common on the boundaries of the SSSI. Water quality data collected on the Levels have supported previous suggestions regarding the decline in the quality of ditch water on the site, a trend that has been especially apparent on the western side of the Levels. This difference has been assigned to inflows provided by the Hailsham STWs, the area of influence of which has been found to be extensive (Jennings, 1994; Section 2.4.8). Biological evidence supports these concerns. Gastropods are particularly vulnerable to drainage and eutrophication and a recent mollusc survey of the Levels has noted the apparent absence or disappearance of *Anisus vorticulus* which had been known in the area for at least 80 years (Killeen, 1994).

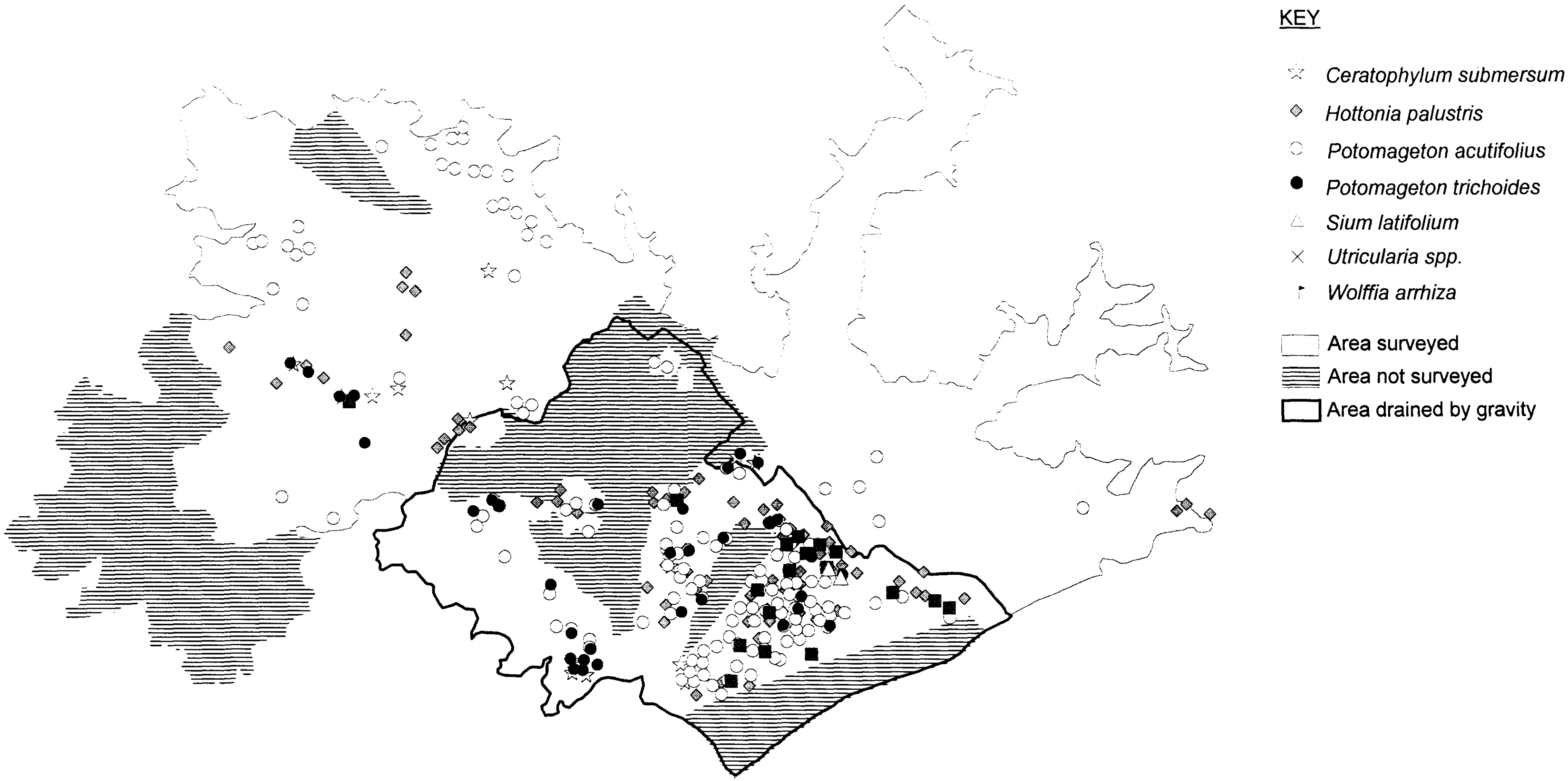


Figure 2.16. Distribution of nationally rare and scarce aquatic floral species on the Pevensey Levels wetland (from Glading, 1986).

(a) (b)

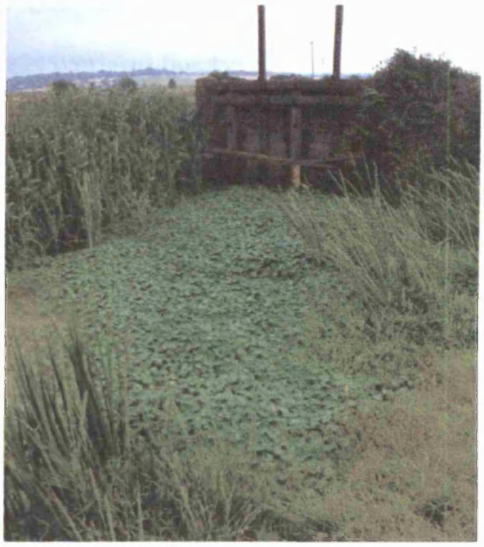


Plate 2.7 . The noxious weeds (a) *Azolla* and (b) *Hydrocotyle rananculoides* (courtesy of Russell Long) growing on pump-drained ditches of the Pevensey Levels.

SIGNATORIES WILL:
• Carry out a staged, approximately six-year cycle of ditch cleaning and re-profiling so as to maintain a constant community of species within the Scheme area.
• Dump, spread and cultivate the spoil from ditch cleaning at least 5 metres from the ditch wherever possible so as to reduce the flow of nutrients back into the ditch and enable safe control of thistles and nettles. These operations will not be allowed before July, as to avoid disturbance to ground-nesting birds.
• Keep water levels as constant as possible: no more than 300 mm below adjoining ground level between January and August and no more than 600 mm below ground level from September to December, subject to there being at least 300 mm of water in each ditch at all times, so as to maximse the water volume available to invertebrates and provide wet pastures for birds.
• Do not apply inorganic or organic fertilisers, lime, herbicides or pesticides (unless specific applications have been previously agreed).
• Graze only at low stocking rates before July in order to avoid nest trampling, maintain permanent pasture and the old marsh contours (low ways and depressions).
• Mow for hay or silage only from July and carry out any topping of thistles or nettles only in July and August.
• Do not carry out any harrowing or rolling after mid-March in order to avoid destroying nests of ground nesting birds.
• Make a record of what management has been carried out, for example, how many head of cattle were on a field for a particular period, or when and where ditch cleaning was carried out.

Table 2.13. Prescriptions associated with the Wildlife Enhancement Scheme on the Pevensey Levels wetland (Reproduced from English Nature, 1990).

2.8. Restoration of the Pevensey Levels wetland

2.8.1. THE WILDLIFE ENHANCEMENT SCHEME

The ecological decline of the Pevensey Levels wetland has been addressed by the introduction of numerous restoration strategies. Parts of the Levels are managed under agreements such as Countryside Stewardship, which has been discussed in Section 1.7.4.2. The areas represented by such agreements is however small, and in terms of the area covered and number of signatories, the most important scheme is the Wildlife Enhancement Scheme (WES)(Figure 2.17). The scheme is managed by English Nature (EN) but funded by the Department of the Environment, Transport and the Regions (DETR). The WES was initially applied only on the Pevensey Levels and the Culm Grasslands SSSI in Devon (EN, 1990), but has since incorporated a variety of habitat types, including heathland (Chris Hewitt, EN, Pers. Comm.). Because of the variety of habitats to which the scheme is applied, prescriptions associated with individual WESs are tailor-made for each site.

On the Pevensey Levels, the WES was launched in November 1991 and, in common with agri-environment schemes reviewed in Chapter One, is a voluntary agreement between the landowner and EN. The scheme offers payments for specific beneficial management operations, including the manipulation of ditch water levels (Table 2.13). The WES attempts to reconcile agriculture and conservation, by promoting a suitable environment which will enable local species to prosper by paying £74 $ha^{-1}yr^{-1}$ to the landowner for farming in a manner which will:

- conserve and enhance the communities of plants and animals living in the freshwater ditches, their banksides and the grasslands,
- maintain the attractiveness of the Levels for the birds that use the area to feed, breed or shelter, and
- continue or reinstate the traditional management of the Levels that has created the conditions in which the special wildlife interests survive (EN, 1990).

Payments are provided twice a year in arrears and agreements run initially for three years. Extra funding is available for some direct works, although to date one of the main criticisms of the scheme by local landowners has been that it does not provide funding for re-engineering gateways that tend to flood due to the higher water levels retained.

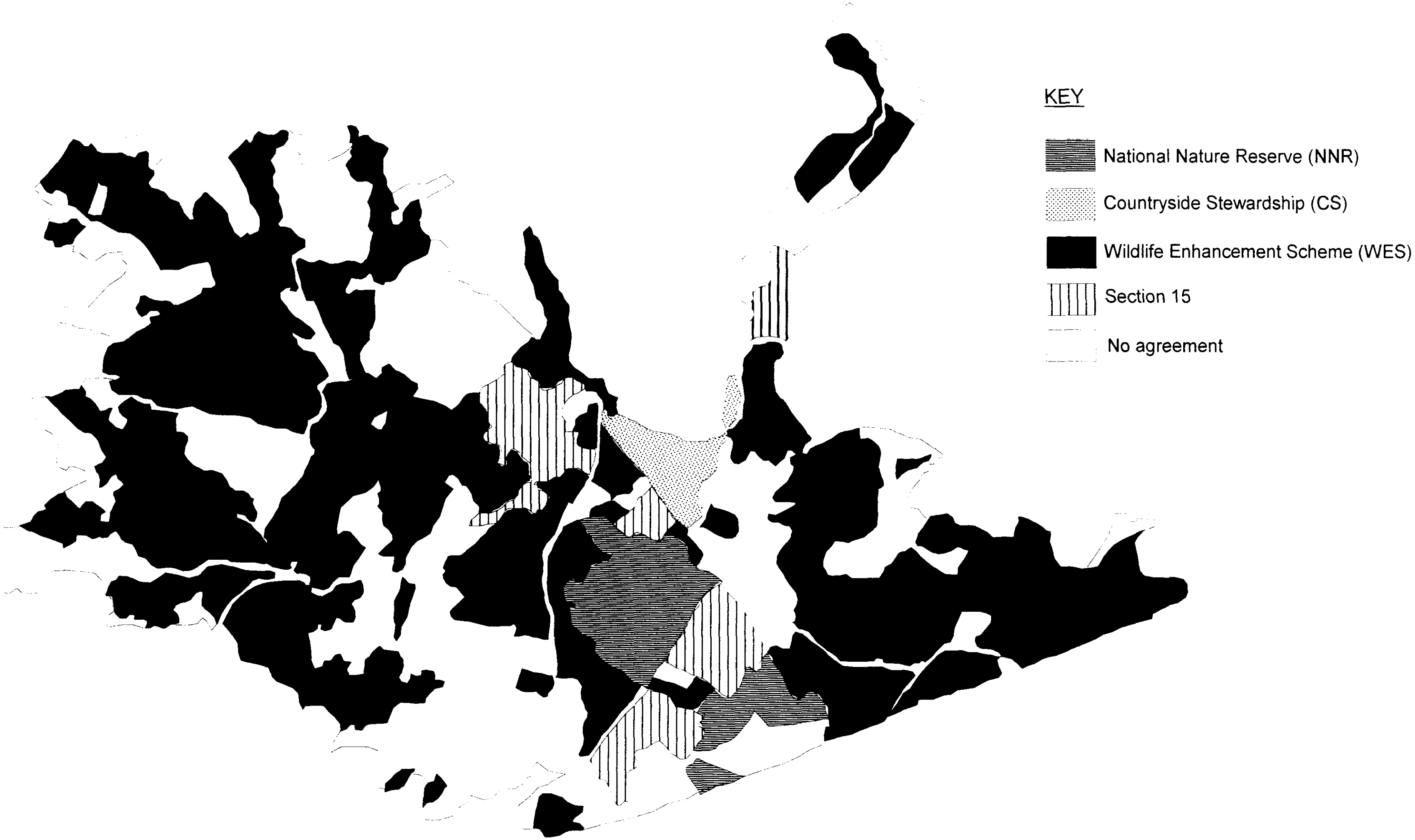


Figure 2.17. Areas within the Pevensey Levels Site of Special Scientific Interest under various management agreements in 1993.

From 1991 to 1995 the scheme was in its 'pilot' stage. Initial uptake of the scheme was considerable. In the first year of the project 60 landowners joined the scheme and by 1993 the WES covered an area of 1671 ha, close to half the area of the SSSI. Combined with other agri-environment schemes, 59% of the SSSI was under some form of management agreement (Hart and Douglas, 1994; Figure 2.17). During this period the economic viability of the scheme, in terms of costs and benefits, was investigated employing contingent valuation methodology, a means of assessing the public's willingness to pay to support the scheme (Willis *et al.*, 1995). The study generated cost-benefit ratios comfortably in excess of unity for the local population and the ratios were considered to be generally positive by MAFF (Douglas and Griffiths, 1995).

In 1998, the WES adopted a new tiered approach more akin to that evident in ESA agreements (see Section 1.7.4.1). Tier One included all WES prescriptions except the water level manipulation option, and Tier Two was instated as the equivalent of the original scheme (Table 2.13). Because water level prescriptions are not included in Tier One, the tier attracts a more limited subsidy than Tier 2 (£31 $ha^{-1}yr^{-1}$ for Tier One as opposed to £74 $ha^{-1}yr^{-1}$ for Tier 2). A number of agreements have also been made between EN and individual landowners, generally where the farmer wants to raise ditch water levels beyond WES prescriptions to flood the land. By the end of 1998, there were 45 landowners in Tier 1, 21 in Tier 2 and six with a mixture of both. Three landowners had entered into individual agreements with EN (Pevensey Levels Study Group, Pers. Comm.). Although this represents an increase in both the number of signatories (75 signatories in 1999 relative to 60 signatories in 1993), as well as in the total areal extent of the scheme relative to 1991 (1863 ha relative to 1671 ha in 1993), the revision of the scheme is likely to have reduced the area subject to higher ditch water levels. In 1991, all 60 landowners would have instated higher levels, while only 21 landowners were in Tier Two of the WES in 1999, although no specific information of the area of land under either tiers is available.

The implementation and continued success of the WES is reliant on solving some of the problems that have been associated with the scheme to date, reviewed in Table 2.14. For example, landowners currently not in WES are mainly intensive dairy and arable farmers. The WES has proved economically viable for traditional beef farmers on the wetland for a number of reasons, but mainly because these landowners have been subject to the greatest fluctuations in market prices for their product during the 1990s. However, landowners on the Levels who own dairy herds or arable farms estimate they would need considerably larger subsidies to participate in the scheme. It has been calculated that a subsidy of £370 $ha^{-1}yr^{-1}$ would be required to make the reversion of arable land to pasture viable in economic terms, with a figure of £500 $ha^{-1}yr^{-1}$ suggested by dairy farmers (Whitbread and Curson, 1992).

Nevertheless, the Pevensey Levels can be considered exemplary in the way in which water level management strategies can be revised for ecological benefit with the consensus of the farming community. Success is generally related to the fact that WES prescriptions do not provide a large variation from current management. Many farmers manage their land along traditional lines and the WES is simply formalising what they already do (Whitbread and Curson, 1992). Steel (1976) reports that during the initial instatement of pumped drainage schemes on the Levels, it was difficult to convince traditional farmers that they could increase their grass production with a low water level regime. Most were sure that a high water level was necessary for 'good' grass, and the water authority and ADAS, who promoted the scheme, had to emphasise that farmers could easily isolate themselves from the system (Steel, 1976).

The most influential factor in the success of the scheme has been the economic circumstances of farming over the period during which the WES has existed. During the last decade, the prices paid for agricultural commodities in the UK have declined (Table 2.15). The Pevensey Levels has been historically geared towards the production of beef (Morton, 1990) and local farm businesses have been severely affected by the Bovine Spongiform Encephalopathy (BSE) and foot-and-mouth crises. During this period farming for wildlife has been increasingly recognised as a viable way to support the farm business, to the extent that subsidies for beneficial management operations can represent up to 10% of the total annual income in local farms (Martin Hole, farmer, Pevensey Levels, Pers. Comm.). The application of WES prescriptions has also allowed some land to be subsequently employed for organic production, since to qualify for Organic Aid no fertilisers must be applied to land for five years. Organic farming is increasing in popularity due to large demand for organic foodstuffs, and probably represents a potential compromise between farmers and nature conservationists on the wetland for the future although at present limited funds are allocated to the Organic Aid Scheme (see Table 1.15).

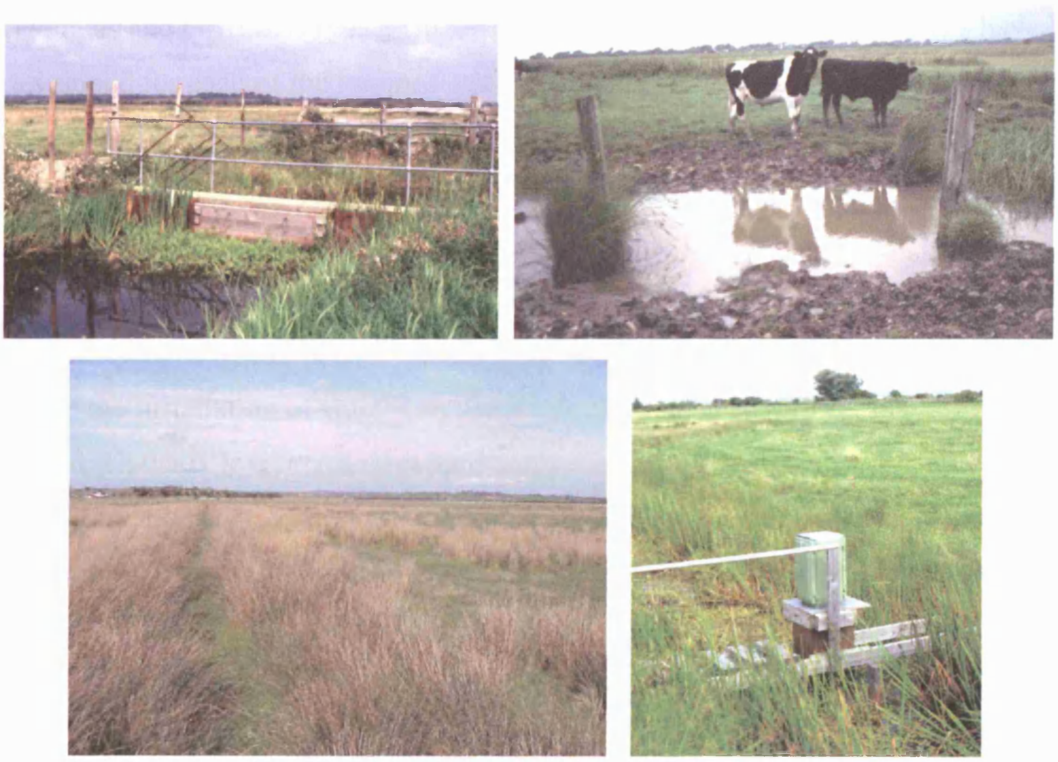
• Rapid land tenure changes are common in areas where the agricultural value of the land is high. As a result WES agreements need to be constantly updated. Indeed to date, the ownership of certain parcels of land on the Levels is still unknown (Basil Lindsey, EN Conservation Officer Sussex, pers. comm.). Since WES is an agreement between the landowner and EN, where land within WES has changed hands EN must negotiate with the new landowner.
• Since the inception of the scheme in 1992 payments per hectare have remained static. Coupled to this, the last few years have seen increases in farming profit margins. This poses problems for continuing land-owner co-operation (Joe Norris, National Farmers Union, Pers Comm).
• Landowners who do not want to adhere to the WES, even where neighbouring land-owners are in the scheme, pose severe problems in terms of prescriptions pertaining to water level management. This land, under ENs obligations, must be hydrologically isolated at great cost.
• Some signatories to the scheme have been unable to maintain water level at the prescription level due to reasons beyond their control. The WES has required the mobilisation of funds to allow the construction of numerous water level management structures. These structures are expensive to build, particularly if their form is to provide flexible water level management. A temporary bund type structure costs about £3000 while a permanent sluice gate can cost in the region of £5000 (Peter Brett Associates, 1997). The problem to date has been locating funds to carry out the works. In most instances these works have been carried out by EN under a *'fixed cost'* agreement which is included in WES or, where a flood defence objective has been justified, the NRA/EA as the Internal Drainage Board have conducted the works. EN however will only provide £2000 yr^{-1} as part of its fixed cost scheme to cover for damage caused by the scheme such as flooded gateways.
• EN has been reluctant to pay for the reconstruction of flooded gateways, a direct result of higher ditch water levels since it is bound to investing only in 'objects for nature conservation gain'. The cost is thus transferred to the landowner. This is an issue regularly raised by farmers on the Levels which poses problems in terms of continuing land owner co-operation. Flooded gateways tend to be low spots in the system and should therefore be included in the initial level survey of the site (Bill Gower, local farmer, pers comm.).
• Results from the pilot area suggest that it is possible for the WES to achieve its objectives with respect to the status of the ditches. However, due to the nature of the soil being predominantly clay, the in-field water table has remained largely unaffected by the raising ditch water levels; water will not permeate through the soil from the ditches (NRA, 1993). In order to create wet areas in the fields, surface ponding of water will be necessary.

Table 2.14. Problems associated with the implementation of the Wildlife Enhancement Scheme on the Pevensey Levels wetland (from discussions between members of the Pevensey Levels Study Group, 1996-1998).

Product	1996	1999
1 kg of Beef	£1.10	£0.80
1 pint of Milk	£0.14	£0.08
1 tonne of Barley	£137.00	£68.00
1 tonne of Wheat	£121.00	£72.00

Table 2.15. Changes in the prices paid for agricultural commodities between 1996 and 1999 (Marion Harding and Bill Gower, Pevensey Levels farmers, pers. comm.).

(a) (b)



(c) (d)

Plate 2.8. Approaches and problems associated with the restoration of the Pevensey Levels wetland. (a) Sluices are used to raise water levels, (b) flooding of gateways due to high water levels, (c) development of rank vegetation, especially *Juncus* spp, (d) hydrological monitoring to predict the likely effects of raising water levels.

2.8.2. WATER LEVEL MANAGEMENT PLANS

A second, more recent mechanism for the restoration of the Pevensey Levels wetland have been Water Level Management Plans (WLMPs; Section 1.7.5). As Internal Drainage Board (IDB) on the wetland, the Environment Agency (EA) is obliged to produce a WLMP for the site under guidelines produced by the Ministry of Agriculture, Fisheries and Food (MAFF; 1994; 1999). On the Pevensey Levels, WLMPs developed have the aim of *'restoring the conservation interest of the site to a level commensurate with its 1990 SSSI designation status'* (Douglas and Griffiths, 1995). It is generally considered that since re-notification there has been no significant overall land use change, therefore to return the site to its re-notification state should require little more than a maintenance of the *status quo.*

A first draft WLMP for the whole wetland was produced by the EA in 1995 (Douglas and Griffiths, 1995). On the Pevensey Levels, individual WLMPs have been developed for each of the pumped sub-catchments on the wetland, as well as for the central gravity area. WLMPs covering the entire wetland area had been developed by the end of 1998 and were modelled on the West Sedgemoor WLMP, which is regarded as a similarly complex site (Basil Lindsey, EN, Pers. Comm.). The schemes *'identify areas where it is believed that water levels could be controlled in order to produce a positive environmental benefit'* (Peter Brett Associates, 1997a-h), and identify locations where the installation of sluices, or the repair of existing structures, may be employed to raise ditch water levels. An important component of the plan is that any new proposed structures should provide sufficient flexibility to allow higher water levels should these be required for new water level management tiers (Douglas and Griffiths, 1995).

WLMPs have been largely complementary to the WES. The objectives of WLMPs have undoubtedly benefited from the existence of WES, not least because WLMPs do not provide subsidies for raising ditch water levels or fund the means by which raised water levels are achieved (Section 1.7.5). As a result of this lack of financial provision, funding for the implementation of WLMPs has been sought elsewhere. In May 2000, the IDB received a £100,000 grant from EN to fund the construction of new sluices on the wetland and by the end of 2003, 400 ha of land could retain ditch water levels close to field surfaces (Mike Porter, Environment Agency, Pevensey Levels Project Officer, Pers. Comm.).

2.8.3. THE PEVENSEY LEVELS STUDY GROUP

One of the main features of the implementation of WES and formulation of WLMPs on the Pevensey Levels has been the high degree of interaction between the different 'user' groups of the wetland, a feature of the hydrological management of the wetland since initial reclamation attempts (Section 2.2.1). Interaction and debate has been possible through the 'Pevensey Levels Study Group', a means of promoting the *'wise use'* of the wetland, which has existed since the inception of the WES in 1991. The importance of this group has been recently recognised by its inclusion in a publication entitled 'Establishing and Strengthening Local Communities' Participation in the Management of Wetlands', part of a worldwide study into stakeholder participation conducted by the Ramsar Bureau (de Sherbinin, 1999).

The Study Group meets approximately twice a year and essentially provides a forum for the discussion of all issues relating to the management of the wetland. Attendees include key figures in the management of the wetland: the local National Farmers Union (NFU) and Farming and Wildlife Association Group (FWAG) representatives, members of the executive of the SWT, Royal Society for the Protection of Birds (RSPB) and EN, as well as EA representatives for Flood Defence, Water Resources and Conservation. Stakeholders present in the study group have frequently asked for best scientific opinion, and as a result, other attendees have included ecologists, hydrologists, geologists and social scientists.

The benefits of this iterative approach have been numerous and span a large variety of topics. In managing the WES, English Nature is used to dealing with individual landowners, but has benefited greatly from the opportunity to speak to the farming community as a whole through the NFU representative. The group has provided the opportunity to disseminate draft WLMPs widely, speeding the application of these on the wetland and ensuring consensus between all stakeholders. Members of the Group have also helped with the formulation of applications to fund the implementation of WLMPs once these have been agreed with local landowners. More recently, the Group has devised a plan for dealing with *Hydrocotyle rananculoides*, of concern to both agricultural and nature conservation stakeholders on the wetland (Section 2.7). The plan, which involved spraying with herbicides, would probably not been publicly accepted had it not been for the credibility provided by the Group's endorsement.

2.8.4. HYDROLOGICAL MONITORING AND MODELLING

Whilst promoting the higher ditch water levels required for nature conservation objectives, the WES does not advocate splash flooding (shallow inundation) of field surfaces. It is therefore considered inadequate by conservationists. A similar criticism can be leveled at WLMPs, as they do not themselves provide an indication of the water level regimes that should be adopted to satisfy objectives. In any case, it is doubtful that given the limited permeability of soils on the wetland (Section 2.3), raising ditch water levels in the ditches to WES prescriptions will increase field water tables to any great extent. As a result of these concerns, in February 1995 a field-scale hydrological monitoring network was installed on the SWT Reserve, one of the WES 'pilot' areas. This complemented previous assessments conducted on the site by Fletcher (1995a; 1995b) discussed in Section 2.4.6. The main objective has been to examine the effects of raising ditch water levels on in-field water table levels and identify the likely sphere of influence of raising levels in a particular ditch (Douglas and Hart, 1994). Three transects combining one metre deep dipwells sunk into the clay, and two-and-half metre deep piezometers monitoring water levels in the peat were installed. Along the transect lines in each field, dipwells were installed at increasing distances outwards from the ditches to the centre of the field (Hart and Douglas, 1994), and each transect was associated with a pen-and-float water level recorder on the adjacent ditch.

In 1995 a tri-partite project was established involving the Environment Agency, University College London and the Institute of Hydrology (now Centre for Ecology and Hydrology, CEH). The objective of this project was to provide a detailed assessment of the hydrological functioning of the wetland, with a particular focus on the evaluation of the sustainability of past, present and future water level management strategies. Based on previous studies conducted on the site, a central aspect of the project involved the consideration of the wetland water balance (Novitski, 1978), applied at a variety of spatial scales. A wide variety of hydrological studies have employed this concept to evaluate wetland functions and the relative importance of individual hydrological processes. Fewer have sought to apply the approach to address management issues. From a nature conservation perspective, the importance of the water balance approach is illustrated by the inclusion of a guide to water balance approach methodology in the recent 'Wet Grassland Guide' (RSPB, ITE and EN, 1997), a publication which represents the *status quo* of wet grassland management strategies in the UK.

The collection of hydrological data from the field-scale monitoring network between 1995 and 1998 was an integral component of this approach. Although in Chapter 3, ditch water level and water table data are mainly employed as a means of evaluating the effects of raising ditch water levels on in-field water table levels, they are also used to evaluate surface and groundwater storage on the wetland. Surface water storage is a particular focus of the catchment-based water balance presented in Chapter 3. On the Pevensey Levels, a large volume of data describing water levels and wetland-wide surface water storage are available but have not been previously applied within a water balance assessment (Section 2.5). Other aspects of the hydrology of the wetland that have not been previously quantified, namely losses to sea, pumping and groundwater storage, are also considered.

Results provided by the wetland water balance are viewed in the context of seasonal and inter-annual climatic variability. The Pevensey Levels are located in one of the driest regions of England and are perceived to have suffered droughts in at least six years since 1989. In particular, data describing water availability at different times of year under different climatic scenarios are employed to evaluate the sustainability of various water level management strategies in water resource terms and promote the *'wise use'* of the wetland. Previous assessments of the water availability wetland-wide have suggested that the area under the influence of wetland restoration schemes associated with higher ditch water levels should be limited due to the scarcity of water resources (Douglas, 1993).

Chapter 5 applies the water balance approach at the field-scale, to develop a hydrological model that considers all the processes effecting water level changes in the wet grassland ditch systems that are the focus of most management strategies in these habitats. The importance of the balance between rainfall and evapotranspiration (Section 2.5), and the limited volume of data describing evaporation relative to rainfall, has also prompted a detailed study of the dynamics of evapotranspiration on the Pevensey Levels. To complement field-scale monitoring, the area chosen for this study has been the Sussex Wildlife Trust reserve. On the reserve, water levels are maintained at higher levels than elsewhere on the wetland (Figure 2.8), allowing an examination of the likely impacts of raising ditch water levels across the wetland on evaporative losses and the wetland water balance in general.

The detailed study of evapotranspiration presented in Chapter 4 was possible due to the on-site availability of equipment enabling the continuous monitoring of actual evapotranspiration, as well as an Automatic Weather Station providing climatic data collected in real time. These data complement climatic data collected at the Horseye climate station (Section 2.4.2), the quality of which have not been previously verified. Results obtained from the evapotranspiration study have been incorporated into both the catchment based and field scale hydrological models described in Chapters 3 and 5 respectively. In this way it has been possible to establish the influence of various evapotranspiration estimates on the accuracy of models describing wetland hydrology, as well as evaluating the importance of evapotranspiration in the hydrological functioning of wetlands.

The thesis concludes with an attempt to address the management conflicts between nature conservation and agriculture evident on the Pevensey Levels wetland, a common management issue in other wet grasslands in the UK (Section 1.6.3). The main difficulties relate to the difficulty of integrating the hydrological requirements of agriculture and nature conservation in the area. Indeed, revised water level management strategies associated with WES and the design of water level management strategies for WLMPs acceptable by all stakeholders are some of the key issues facing the future management of the Pevensey Levels wetland. In Chapter 6 the field-based hydrological model is employed to evaluate the impacts of various ditch water level management regimes on various stakeholders on the wetland. For this purpose, the water level requirements of these stakeholders have been investigated. The water level requirements of selected flora and fauna of wet grassland, and those of the farming community, are presented in Chapter 6. The development of a hydro-ecological tool to predict the impacts of various scenarios on wetland stakeholders is presented with a view towards identifying water level management strategies that may lead to the 'wise use' of the site with the consensus of all wetland stakeholders.

CHAPTER 3

WATER BALANCE OF THE PEVENSEY LEVELS:
VIABILITY OF CURRENT AND PROPOSED WATER LEVEL MANAGEMENT STRATEGIES

3.1. Introduction

A large volume of data describing the hydrological functioning of the Pevensey Levels wetland are routinely collected by the operating authority, the Environment Agency (EA). Data provided by this hydro-meteorological monitoring network are employed primarily for real-time flood defence purposes. As a result, to date, few data have been compiled and employed for the computation of an integrated catchment-based, water balance considering all inflows, outflows and sinks on the wetland. Water balance assessments conducted to date have been reviewed in Section 2.5., but all have omitted crucial aspects of the wetland water balance, namely losses to sea and surface water storage on the wetland. There is an urgent need for water-resource managers to consider all the parameters of the water budget (Rushton, 1996). This is certainly the case on the Pevensey Levels, where the introduction of the Wildlife Enhancement Scheme (WES) and Water Level Management Plans (WLMPs), raises some important water resource issues.

The WES and WLMPs both advocate an increase in water levels on a wetland wide basis, but the availability of water for these schemes has not been fully considered. Additionally, the effects of increases in water availability on evaporative losses from the wetland need to be considered. These are crucial requirements in order to assess the sustainability of revised water level management strategies in the area, a requirement which provides the justification for the application of the water balance approach presented in this chapter. Issues of sustainability also need to be considered in the context of inter-annual climatic variability and longer term climate change, which are generally expected to reduce rainfall in temperate regions (Hulme and Barrow, 1997). In the local area, at least six of the years between 1988 and 2004 have been subject to perceived droughts (1989-1992, 1995, 1996, 2003) pushing the issue of climate change higher up the agenda of local wetland and water resource managers.

In considering the water balance of the Pevensey Levels wetland, this Chapter also provides an assessment of the hydrological functioning of the wetland at the catchment scale. Data describing the hydrological processes effecting the wetland water balance are reviewed and trends are identified that help explain, and predict inter-annual and seasonal trends in wetland water availability. Based on the preliminary analysis of the drainage system of the Pevensey Levels reported in Chapter 2, three distinct spatial scales can be identified relative to wetland hydrological functioning: the wetland-wide scale, the pump-unit scale and the field scale. Although the organisation of the drainage system is such that these scales are inter-linked, this chapter considers data gathered at these three distinct scales individually as a means of simplifying the description and construction of a conceptual model of wetland hydrological functioning.

The water balance assessment is employed mainly to address water resource issues highlighted by local stakeholders. This includes consideration of the availability of water for a variety of water level management schemes, and the sustainability of surface water abstraction for public water supply from the wetland. Abstraction has been found to be significant in dry summers (Section 2.5) and as an outflow to the system, may be important in terms of the availability of water for revised water level management strategies at crucial times of year.

This Chapter also considers methodological issues associated with the application of the wetland water balance approach. For example, the recently published 'Wet Grassland Guide' (RSPB *et al.*, 1997) suggests the use of rainfall, evaporation and soil moisture deficit data (SMD) as sufficient to provide an indication of wetland water availability and the validity of this approach is considered in the context of the Pevensey Levels. Although for most features of the hydrology of the Pevensey Levels, data were available from about 1970, the hydrological data employed in the water balance assessment considers the period between 1995 and 1998. This is the period when hydrological data describing all the components of the water balance are available, especially those describing field scale hydrology. Nevertheless, this chapter provides a basis for retrospective water balance calculations to be conducted beyond the period presented, because the catchment water balance has been operationalised as a simple Excel water balance model.

In considering the wetland water balance, the model provides an account of the hydrological functioning of a typical wet grassland wetland at the catchment scale. Understanding of the hydrology of these wetland types has generally been limited to studies of their soils and site specific guidelines (Cook and Moorby, 1993), although some detailed accounts of hydrological functioning at the field scale have been provided by Armstrong (1993) and Youngs (1989) (Section 1.6.5.). Thompson and Hollis (1993) and Gilman (1994) have conducted studies similar to that presented in this chapter on the North Kent Marshes and the Somerset Levels respectively, both flagship sites in terms of nature conservation importance. Probably, the most comprehensive evaluation of wet grassland hydrological functioning to date has been provided by a study commissioned by Hydraulics Research, Wallingford. As stated in Section 1.6, an important feature of wet grasslands is their association with drainage systems managed for agricultural and flood defence purposes, and the publication entitled 'The Hydrology and Hydraulics of Pumped Drainage Systems' (Samuels, 1993), has provided substantial support to the development of methods presented in this chapter. This chapter aims to complement these studies, providing a review of the hydrology of the Pevensey Levels wetland.

3.2. The water balance of the Pevensey Levels

For any hydrological system, and based on the principle of mass conservation, the wetland water balance can be expressed as

Inflow = Outflow +/- ΔStorage +/- Error (*e.g.* Gilman, 1989) (Equation 3.1)

For the Pevensey Levels, based on previous work by Douglas (1993) and Loat (1994) reviewed in Chapter 2, this can be expressed as:

$$\text{Inflows} = P + Q_{\text{Wallers Haven}} + Q_{\text{STW}} \qquad \text{(Equation 3.2)}$$

$$\text{Outflows} = E + Et + A + Q_{\text{sea}} \qquad \text{(Equation 3.3)}$$

and

$$\text{Storage} = S_{\text{Surface water}} + S_{\text{Water Table}} + S_{\text{Soil}} \qquad \text{(Equation 3.4)}$$

where	P	is precipitation,
	E	is evaporation from open water surfaces,
	Et	is evapotranspiration from vegetated surfaces,
	$Q_{\text{Wallers Haven}}$	is Wallers Haven discharge,
	Q_{STW}	is discharge from Sewage Treatment Works,
	Q_{Sea}	are losses to sea,
	A	is abstraction for public water supply,
	$S_{\text{Surface water}}$	is surface water storage in ditches, pumped and embanked channels,
	$S_{\text{Water Table}}$	is shallow groundwater storage and
	S_{Soil}	is the soil moisture storage.

A history of intensive management for agriculture through direct intervention with local hydrology, has left an extensive hydro-meteorological monitoring network on the wetland that allow the estimation of the magnitude of each of these processes at the catchment level. The hydrological monitoring network consists of a variety raingauges, stage boards and water level recorders shown in Figure 3.1.

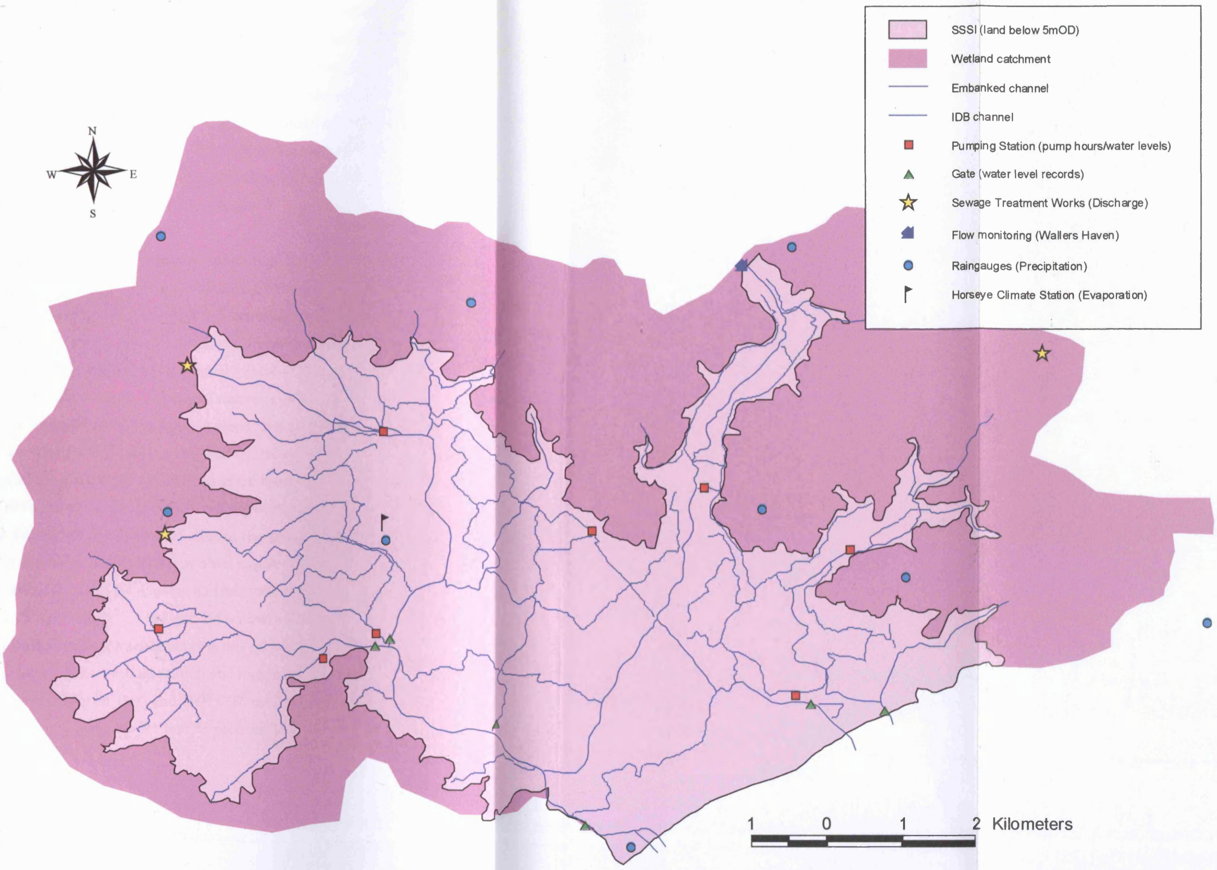


Figure 3.1. The hydrometric network employed for the calculation of the water balance of the Pevensey Levels wetland.

The various data sets required to develop a hydrological assessment considering all the inflows, outflows and sinks on the wetland were only coincident between January 1995 and December 1998 so that the water balance study was limited to this period. The same period has been used for other hydrological studies presented in this thesis, such as the modelling studies described in Chapters 5 and 6. The use of this period also allowed the confidence in the application of the detailed evapotranspiration data gathered during the summers of 1996 and 1997 within the wetland water balance. Results provided by the evapotranspiration studies, including their relevance to the wetland water balance are considered in Chapter 4.

The climatic conditions evident between January 1995 and December 1998 afforded further support for the choice of period. Climatic conditions throughout this period were characterised by considerable inter-annual variability, allowing water resource assessments to be conducted under different scenarios of water availability. Relative to the 1961-1990 long-term annual rainfall at Horseye (763 mm), 1995 and 1996 were drier (679 mm and 592 mm of rain respectively), and 1997 and 1998 were wetter than average (901mm and 856mm respectively) (Figure 3.2). In 1995 and 1996 only four and three months respectively had rainfalls above respective 1961-1990 monthly averages, compared with six and seven months in 1997 and 1998. The years of 1995 and 1996 can therefore be considered representative of the perceived climatic conditions that current predictions of climate change suggest. In contrast, in both 1997 and 1998 of particular importance was a wet month of June, a period of traditionally high evaporative demand. As a result, soil moisture deficits provided by MORECS (SMD_{MORECS}) were higher in 1995 and 1996 than in 1997-1998 (Figure 3.2). Evaporation was similarly variable during this period, although there was no relationship between low annual rainfall and high annual tank evaporation. Tank evaporation was higher in 1995 and 1997 (788 mm and 714 mm respectively) than in 1996 and 1998 (677 mm and 691 mm respectively).

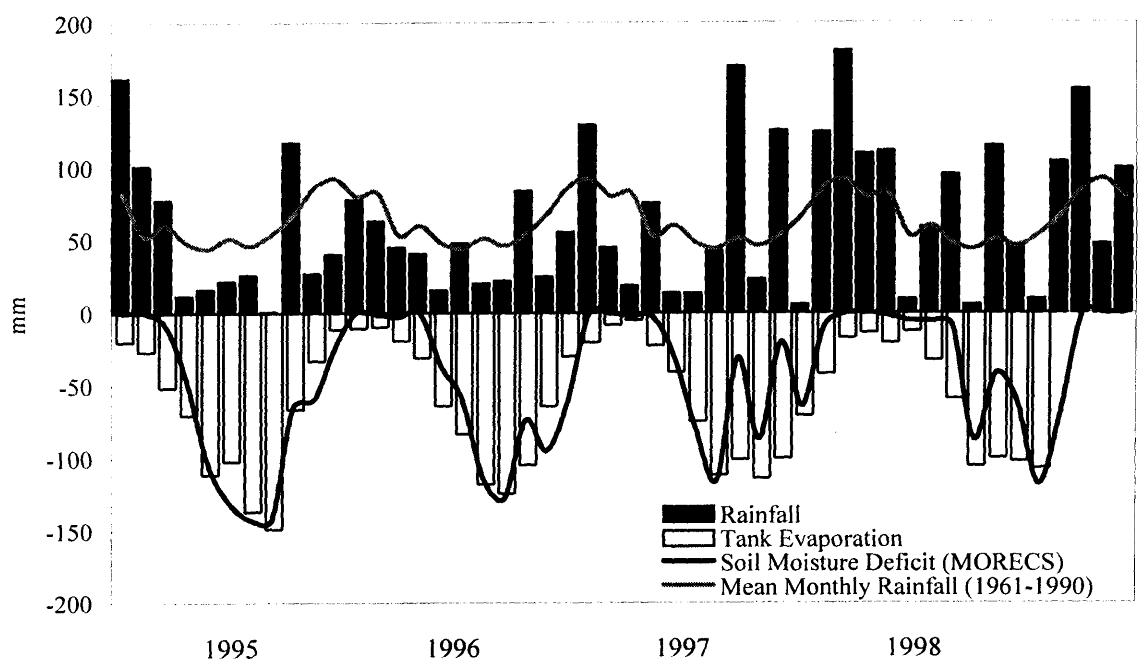


Figure 3.2. Monthly Horseye rainfall and tank evaporation, and MORECS Soil Moisture Deficit [SMD], January 1995 - December 1998.

3.3. Catchment hydrology

3.3.1. RAINFALL

3.3.1.1. Data availability

Of all the components of the hydrological cycle rainfall is the most commonly measured (Shaw, 1993). In the UK there are about 4500 raingauges, giving an average density of approximately one gauge per 60 km^2 (Ward and Robinson, 1989). On the Pevensey Levels, a history of intensive hydrological management for agriculture has resulted in this figure being approximately one raingauge per 7 km^2. The existence of numerous raingauges within the wetland catchment was especially advantageous in terms of the accuracy of the water balance assessment. Although previous studies have identified the spatially variability of rainfall over the wetland (see Section 2.4.2), water balance studies conducted by Douglas (1993) and Loat (1994) have considered rainfall at Horseye to be descriptive of the entire wetland region. The water balance assessment presented in this chapter thus improves the accuracy of previous water balance studies by providing a truly catchment based approach, and adopting a more spatially distributed approach to the calculation of rainfall than has been previously possible.

To determine the raingauges representative of wetland conditions, the catchment boundary was defined from elevation contours obtained from Ordnance Survey maps of the area. The hydrological catchment boundary defined is shown in Figure 3.1. To the east, the Pevensey Levels are bound by the catchment of the Cuckemere River, to the west by the catchment of the Coombe Haven. The northern boundary is formed by the essentially upland catchments which converge upstream of Boreham Bridge to form the Wallers Haven (Section 2.4.1). The total catchment area of the Pevensey Levels wetland was calculated as 56.7 km^2 , extensively larger than the value of 35 km^2 employed by Douglas (1993) and Loat (1994) relating to the SSSI only, with potentially important implications for the assessment of water availability at the wetland scale. Catchment delineation allowed the identification of nine raingauges either within, or on the boundaries of, the hydrological catchment of the wetland (Figure 3.1). However, only those at Horseye and Pevensey Bay could be said to describe 'lowland' rainfall, since the remaining gauges are all located at altitudes in excess of 10 m OD. A review of the characteristics of all the raingauges located within the wetland catchment has been previously given in Table 2.4.

All raingauges within the catchment boundary were part of the Environment Agency's rainfall monitoring network. Raingauges are of the standard Met Office markII type and are measured on a daily basis at 0900 (Russell Long, Environment Agency, Pers. Comm.). Two of the gauges employed were tipping bucket devices, known to underestimate rainfall during intense storms and therefore not necessarily comparable to storage gauges. Nevertheless, the constant slope provided by double mass analysis (Wilson, 1979) between Horseye rainfall and all other raingauges, including tipping bucket devices, suggested that the data were of good quality and comparable. For raingauges on the wetland, the results of the double-mass analysis are shown in Figure 3.3. The representative area associated with each raingauge was calculated by application of the Thiessen polygon method to the raingauge network. The Thiessen polygon method weights the fractions of a catchment area represented by each raingauge by dividing the area into polygons by lines that are equidistant between pairs of stations (Shaw, 1993). For raingauges on the Pevensey Levels, Table 3.1 lists the representative areas associated with each gauge. These data were those applied to the raingauge data to evaluate the contributions of rainfall to the Pevensey Levels wetland.

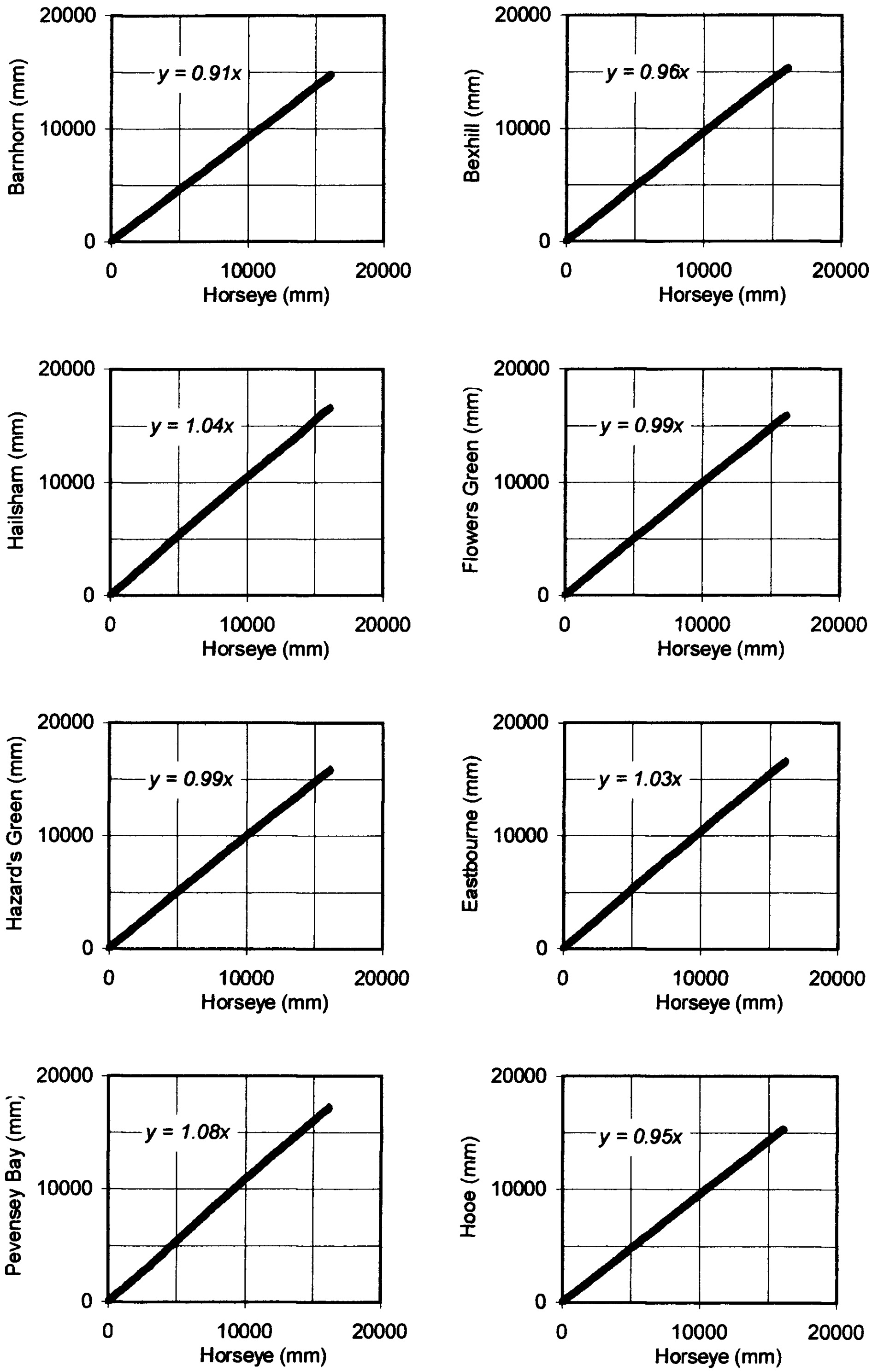


Figure 3.3. Double mass analysis (daily data 1970-1998) of Horseye rainfall relative to other raingauges employed in the water balance assessment.

3.3.1.2 Trends

For all stations within the wetland catchment, mean monthly rainfall maxima (based on the 1961-1990 mean monthly rainfall) occurred in November, followed by October, December and January respectively (Table 2.4). Minima were in May, except for the Hooe gauge, with minima in July, which was the second driest month for all other raingauges, followed by April and June (Table 2.4). Of all the available records, the highest mean annual rainfall was that recorded at Hailsham (Table 3.1). Mean annual rainfall was generally larger for stations in the west than those to the east and on an annual basis, rainfall at the easternmost point of the catchment was approximately 8% less than that on the western boundary. Figure 3.4 shows rainfall isohyets for the Pevensey Levels calculated from the mean annual rainfall between 1961 and 1990, shown in Table 3.1. The isohyets provided closely replicate those previously provided by the Southern Water Authority (1973).

The longevity of some of the raingauge records provided the opportunity to examine long-term rainfall trends in the area, providing a hydrological context to the period chosen for the water balance assessment. Three raingauges in the local area (Pevensey Bay, Hazard's Green and Bexhill) have records pre-dating 1940. The first rainfall data at Eastbourne was collected in 1886. Although based on the Thiessen polygon approach both the Eastbourne and Bexhill gauges were located outside the area of influence of the wetland, the length of the records from these sites supported their use to evaluate long term rainfall trends. Analysis of all raingauge records relative to respective 1961-1990 mean monthly rainfall supported the perception that the 1990s have been characterised by a period of drought equivalent to that of the mid 1970s. Roughly cyclical variations in monthly rainfall relative to respective 1961-1990 means were apparent in all records, with the mid-1970s and early 1990s located in the troughs of this cyclical pattern (Figure 3.5). The Eastbourne record also allowed the identification of other notable droughts during the 1900s, 1940s and the early 1920s (Figure 3.6.a). However, based on the application of the method employed by Bromley *et al.* (1997) for the analysis of long-term rainfall trends, available rainfall data suggested that it has been getting generally wetter throughout this century (Figure 3.6.b.). Results supported suggestions by Douglas (1993) (Section 2.4.2), although trends could not be verified due to the lack of information regarding the history of the Eastbourne gauge.

Raingauge	Period of record	Annual Mean (mm)	Maximum (mm)	Minimum (mm)	Representative Area (km^2)
Pevensey Bay	1931 - 1998	782	1079	526	5.2*
Eastbourne	1886 - 1998	790	1178	400	0.0
Hazards Green	1931 - 1998	763	1023	517	3.2*
Horseeye	1968 - 1998	763	1026	529	11.3*
Hailsham	1962 - 1998	792	1071	517	8.7*
Bexhill	1931 - 1998	729	1043	510	1.3*
Barnhorn	1967 - 1998	690	930	484	8.5*
Flowers Green	1972 - 1998	755	1029	533	8.7*
Hooe	1967 – 1998	722	946	484	9.8*
Hellingly	1937 - 1998	766	1130	533	0.0

Table 3.1. Details of raingauges around the Pevensey Levels wetland (*used in wetland water balance calculations). Representative areas have been calculated using the Thiessen polygon approach.

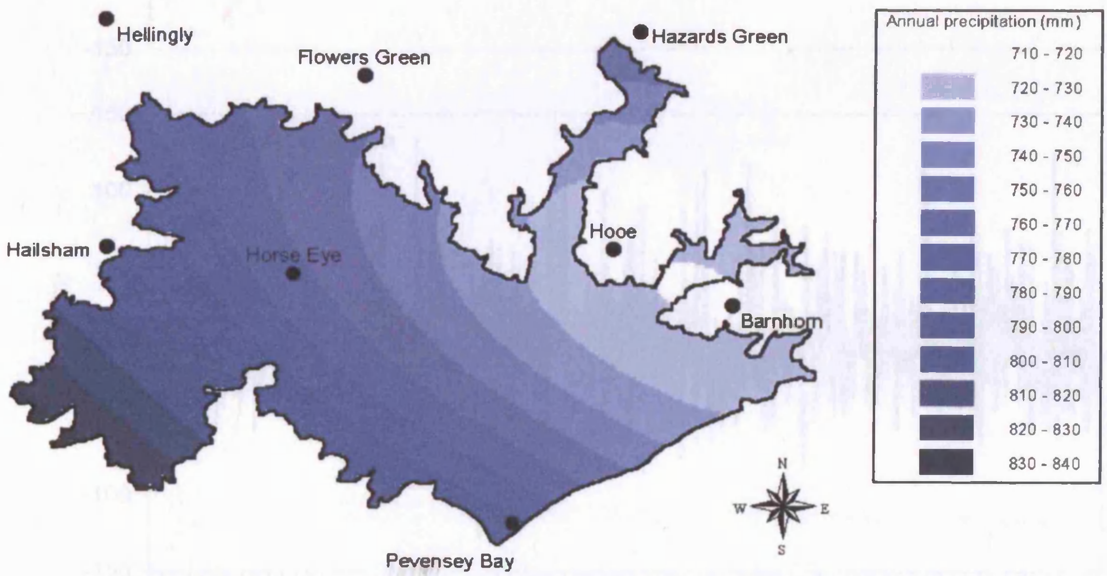


Figure 3.4. Isohyets for the Pevensey Levels SSSI. Calculated from mean annual rainfall 1961-1990 (see Table 3.1).

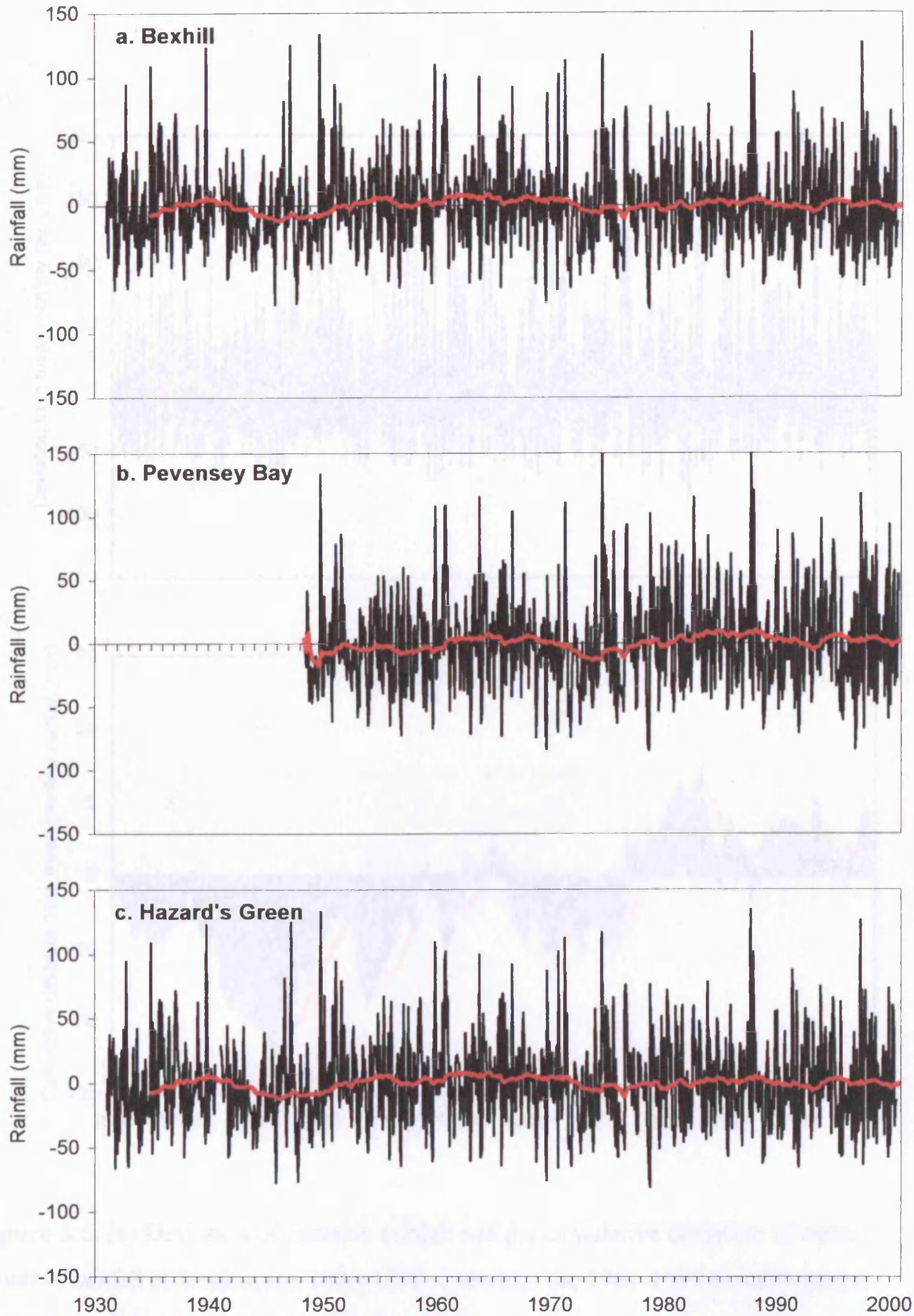
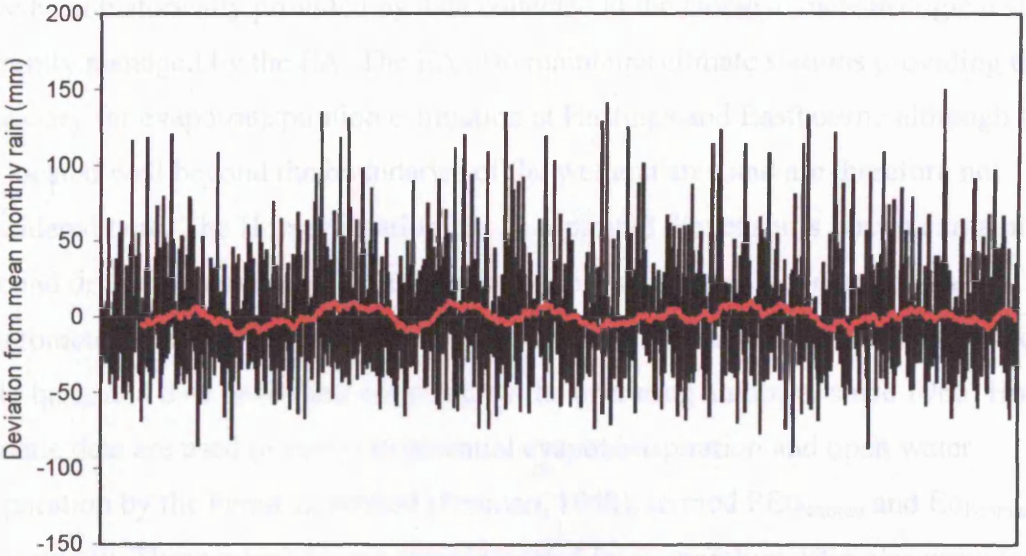


Figure 3.5. Departure of monthly rainfall 1970-1998 for raingauges used in water balance calculations from respective long-term monthly means. The red line indicates the 24-month running mean.

(a)



(b)

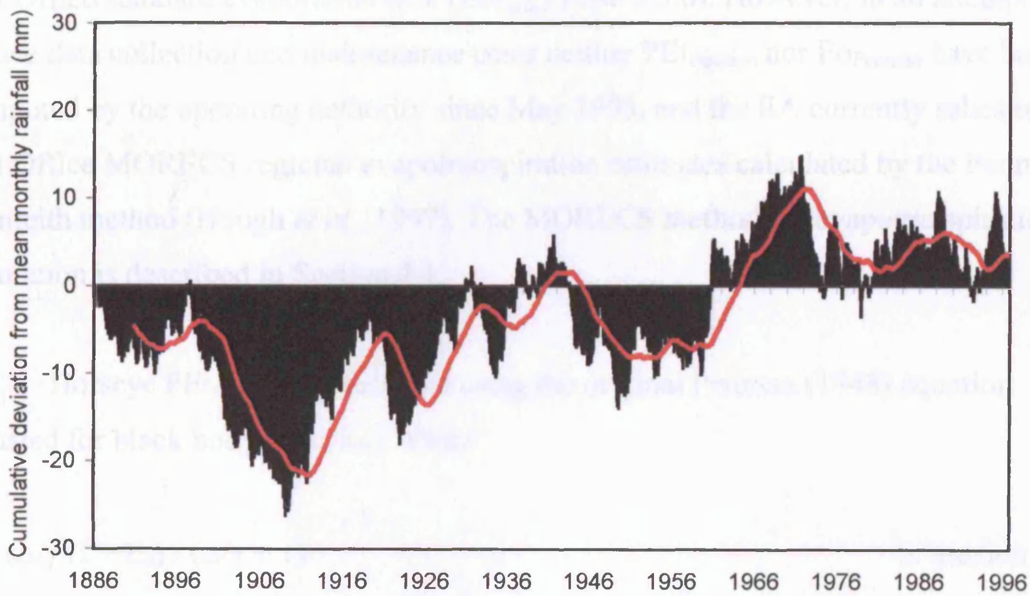


Figure 3.6. (a) Deviation of monthly rainfall and (b) cumulative deviation of mean monthly rainfall at Eastbourne 1886-1998 relative to the 1961-1990 monthly means. The red line indicates the 5-year running mean.

3.3.2. EVAPORATION AND EVAPOTRANSPIRATION

3.3.2.1. Methods and trends

Routine evaporation and evapotranspiration estimates for the Pevensey Levels wetland have been historically provided by data collected at the Horseye meteorological station, currently managed by the EA. The EA also maintains climate stations providing the data necessary for evapotranspiration estimation at Hastings and Eastbourne although these are located well beyond the boundaries of the wetland area and are therefore not considered here. The Horseye station has an aspirated Stevenson's screen containing wet and dry bulb thermometers, a Campbell-Stokes sunshine recorder and an anemometer mounted at 2m height (Plate 2.5.a). All instruments are read at 0900 on a daily basis and data have been collected by the operating authority since 1966. Horseye climatic data are used to compute potential evapotranspiration and open water evaporation by the Penman method (Penman, 1948), termed PEt_{Penman} and Eo_{Penman} respectively. These estimates are complemented by evaporation estimates provided by a Met Office standard evaporation tank (Eo_{Tank})(Plate 2.5.b). However, in an attempt to reduce data collection and maintenance costs neither PEt_{Penman} nor Eo_{Penman} have been computed by the operating authority since May 1995, and the EA currently relies on Met Office MORECS regional evapotranspiration estimates calculated by the Penman-Monteith method (Hough *et al.*, 1997). The MORECS method for evapotranspiration estimation is described in Section 4.3.

Horseye PEt_{Penman} is calculated using the original Penman (1948) equation adjusted for black body radiation, where

$$E = (\Delta/\gamma\ H + E_a) / (\Delta/\gamma + 1) \qquad \text{(Equation 3.5.)}$$

and

$$H=0.75\ R_a(0.16+0.62\ n/N) - 0.95\sigma T^4\ (0.47 - 0.075\ e_d)\ (0.17 + 0.83\ n/N) \qquad \text{(Equation 3.6.)}$$

$$E_a = 0.35\ (e_a - e_d)\ (1 + u_2 / 160.9) \qquad \text{(Equation 3.7)}$$

where

Δ is the slope of the saturated vapour pressure curve for water at mean air temperature T (in mm Mercury),

γ is the constant of the dry and wet bulb psychrometers,

σ is the Steffan Boltzmann constant,

T is temperature (in ^{o}C),

e_a is the vapour pressure at dew point T,

e_d is saturated vapour pressure at mean air temperature (mm Mercury),

u_2 is mean daily wind run at two metres,

R_a is a function of latitude and month of the year,

n is actual sunshine hours and

N is theoretical sunshine, also based on latitude and time of year.

Values for n, T ($T = T_{max} + T_{min} / 2$) and u_2 are provided by data from Horseye and values for R_a, N, 0.95 σT^4, e_a and e_d are taken from tables provided by the Ministry of Agriculture, Fisheries and Food Technical Bulletin No. 16 (1967). For the calculation of Eo_{Penman} the same equation is employed, with adjusted multipliers for the aerodynamic (Ea) and heat storage (H) functions of equations 3.6 and 3.7. To account for the different albedo and aerodynamic properties of a water surface, equation 3.6 becomes

$$H= 0.93R_a\,(0.16+0.62\ n/N) - 0.95\sigma T^4\,(0.47-0.075e_d)\,(0.17+0.83\ n/N) \quad \text{(Equation 3.9)}$$

and equation 3.7 becomes

$$E_a = 0.35\,(e_a - e_d)\,(0.5 + u_2 / 160.9) \quad \text{(Equation 3.10)}$$

Equation 3.8 is similarly used to convert Horseye windspeed data to the equivalent wind run at two metres and a set of tables for values of R_a, N, 0.95 T^4, e_a and e_d for the calculation of Eo_{Penman} are also available in the MAFF Technical Bulletin no. 16.

The spatial variability of evaporation and evapotranspiration over the wetland could not be examined because Horseye is the only station within the wetland catchment providing the data necessary for the estimation of evapotranspiration and evaporation. Application of the Thiessen polygon approach illustrated that only the Horseye station could be considered representative of conditions on the wetland. Nevertheless, the availability of data at Eastbourne allowed some comparative assessments. Double mass analysis of PEt_{Penman} and Eo_{Penman} data from the Eastbourne and Horseye stations (Figure 3.7) suggested that data were of good quality and comparable, illustrated by the constant slope of the relationship between both variables, and the low degree of scatter apparent in the relationships. In the case of both PEt_{Penman} and Eo_{Penman} rates at Eastbourne tended to be higher than those at Horse Eye (Table 3.2), in accordance with suggestions by the Southern Water Authority (Section 2.4.2).

3.3.2.3. Calculating evaporative losses from the Pevensey Levels wetland

To estimate losses by evaporation and evapotranspiration from the wetland, the method used in previous studies by Douglas (1993) and Loat (1994) was initially employed. Losses from the wetland by evapotranspiration and evaporation were determined by calculating the area of land and open water and taking PEt_{Penman} and Eo_{Tank} as representative of each of these areas respectively. For the area of open water, the value provided by Douglas (1993) and Loat (1994) was employed (232,439 m^2) and the land area was taken as 56.4 km^2, the total catchment area excluding the open water area. Problematically however, PEt_{Penman} data were only available until May 1995. Horseye Eo_{Tank} were the only available estimates for the entire water balance period. To represent ET from grass surfaces on the wetland, Eo_{Tank} measurements were multiplied by a suitable coefficient developed by analysis of local historical data. This approach was necessary because tank devices are known to be unreliable for estimating evapotranspiration from vegetated surfaces, because being small, shallow, unpainted, galvanised iron containers they are unable to replicate evapotranspiration from a vegetated surface and tend to over-estimate wetland evaporation (Rushton, 1996).

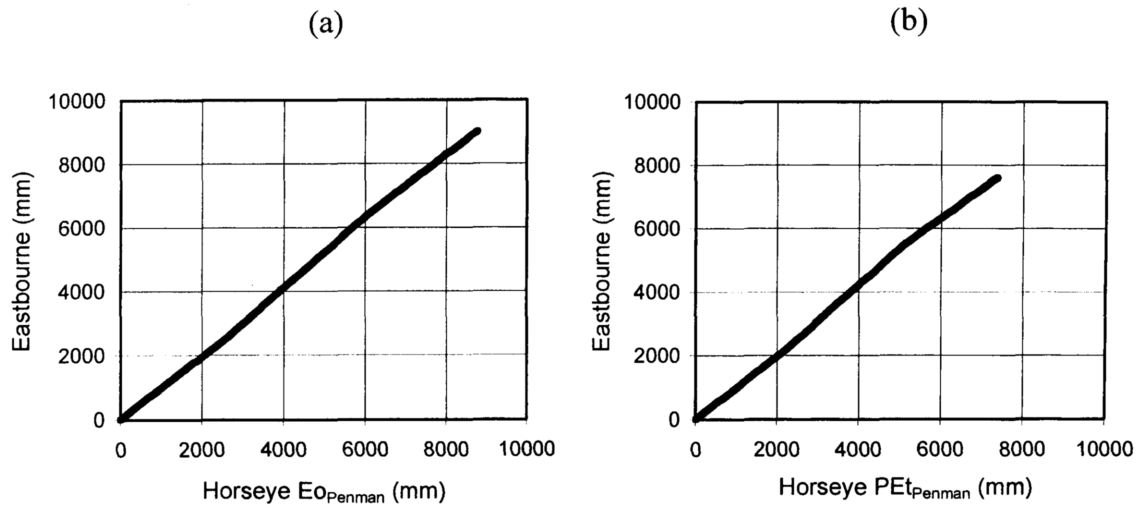


Figure 3.7. Double mass analysis of daily Eastbourne and Horseye (a) Eo_{Penman} and (b) PET_{Penman} estimates for the period between 1975 and 1989.

	Eastbourne		Horseye		
	Eo_{Penman}	PEt_{Penman}	Eo_{Penman}	PEt_{Penman}	Eo_{Tank}
January	7.8	8.2	10.8	11.2	14.3
February	13.9	12.1	15.4	13.4	14.9
March	36.6	30.6	37.6	31.8	35.1
April	68.6	57.2	67.6	56.5	61.0
May	103.3	85.4	100.5	83.0	90.7
June	116.7	97.7	112.2	93.0	102.8
July	120.6	100.9	116.5	96.9	109.2
August	105.2	87.6	98.5	80.5	95.0
September	63.6	53.1	58.4	47.2	57.9
October	31.6	27.2	29.6	24.9	33.5
November	12.1	11.8	12.8	12.1	19.1
December	8.6	9.6	10.7	11.6	13.2
ANNUAL	688.6	581.5	670.6	562.2	646.6

Table 3.2. Mean monthly evaporation for Horseye and Eastbourne between 1970 and 1989.

Coefficients for the estimation of PEt_{Penman} were developed by correlation between the different evaporation and evapotranspiration estimates available at Horseye. The relationship between Horseye estimates and MORECS potential evapotranspiration (PEt_{MORECS}) and actual evapotranspiration (AEt_{MORECS}) was also investigated to assess the validity of the EA approach of relying on regional estimates to calculate wetland evapotranspiration. Figure 3.8 shows the results of these comparisons and results are summarised in Table 3.3 as a correlation matrix describing the slope of the relationship between each evaporation/evapotranspiration estimate. In each case the values presented are the coefficients that best describe the relationship between the estimates and can be used to predict one from the other. The accuracy of the coefficient in each case is quantified by the value of the coefficient of determination (R^2), which for each relationship is given in brackets in Table 3.3.

At Horseye, Eo_{Tank} underestimated Eo_{Penman}, but overestimated PEt_{Penman}. Differences between Eo_{Penman} and PEt_{Penman} relative to Eo_{Tank} could be attributed to the different multipliers used in the energy budget component of equations used to calculate the potential evapotranspiration and open water evaporation respectively. Due to the higher multiplier used in the energy budget equation for Eo_{Penman} (Equation 3.9), evaporation estimated by this method will always exceed estimates of PEt_{Penman} (Equation 3.6) when radiant energy is the process governing evaporation (mainly during the summer months). In contrast, when wind energy dominated the process, the reverse was the case. During the winter months Eo_{Penman} commonly exceeded PEt_{Penman} due to the higher multiplier used in equation 3.7 relative to 3.10.

Due to the lack of potential evapotranspiration estimates for the water balance period, of particular interest were the results of the correlation between Horseye PEt_{Penman} and Horseye Eo_{Tank}, the only estimate available for the entire period chosen for the water balance assessment. The coefficient obtained for the calculation of PEt_{Penman}, 0.88 Eo_{Tank}, was equivalent to the factor of 0.88 advocated by Kadlec (1989) for use in 'small, diked wetlands' and close to the value of 90% of standard open-field pan evaporation used as an approximation of the potential evaporation used by Riekirk and Korhnak (2000) on the Florida Everglades. Furthermore, the limited degree of scatter evident in the relationship (R^2= 0.97) suggested that the calculation of PEt_{Penman} from Eo_{Tank} would not introduce significant inaccuracies to the water balance calculation. Chapter 4 considers the suitability of this method in more detail.

The relationship between MORECS potential evapotranspiration estimates (PEt_{MORECS}) and Horse Eye PEt_{Penman} was close to unity and characterised by a small degree of scatter, supporting the Environment Agency policy of relying on regional estimates to reduce data collection costs. However, none of the Horseye estimates could be used to estimate actual evapotranspiration calculated by the MORECS method (AEt_{MORECS}) accurately. Relationships between this and other estimates were characterised by an extensive degree of scatter, especially at the highest rates of evaporative demand (see Figure 3.8). This result was attributed to the assumption made by all potential evaporation estimates regarding an unlimited water supply. Their application therefore becomes hazardous when the vegetation experiences water stress, since the rate of actual evapotranspiration is influenced by the restricted supply of water from the soil (Wallace, 1991). A method for the calculation of AEt from Horseye E and Et estimates and its influence on water balance calculations is considered in Chapter 4, but for the initial water balance calculation, a value of 0.88 Eo_{Tank} was adopted to replicate the method advocated by Douglas (1993) and Loat (1994).

	Eo_{Penman}	PEt_{Penman}	Eo_{Tank}	PEt_{MORECS}	AEt_{MORECS}
Eo_{Penman}		0.83 (0.99)	0.93 (0.97)	0.81 (0.95)	0.61 (0.49)
PEt_{Penman}	1.20 (0.99)		1.13 (0.97)	1.00 (0.96)	0.73 (0.49)
Eo_{Tank}	1.06 (0.97)	0.88 (0.97)		0.87 (0.96)	0.64 (0.45)
PEt_{MORECS}	1.21 (0.95)	1.00 (0.96)	1.14 (0.96)		0.75 (0.60)
AEt_{MORECS}	1.43 (0.49)	1.19 (0.49)	1.33 (0.45)	1.19 (0.60)	

Table 3.3. Regression matrix of different evaporation and evapotranspiration data available on the Pevensey Levels wetland. Based on correlation of monthly data for the period between 1970-1994 forced through 0,0. Values indicate the slope of the relationship. The coefficient of determination of each relationship is given in brackets.

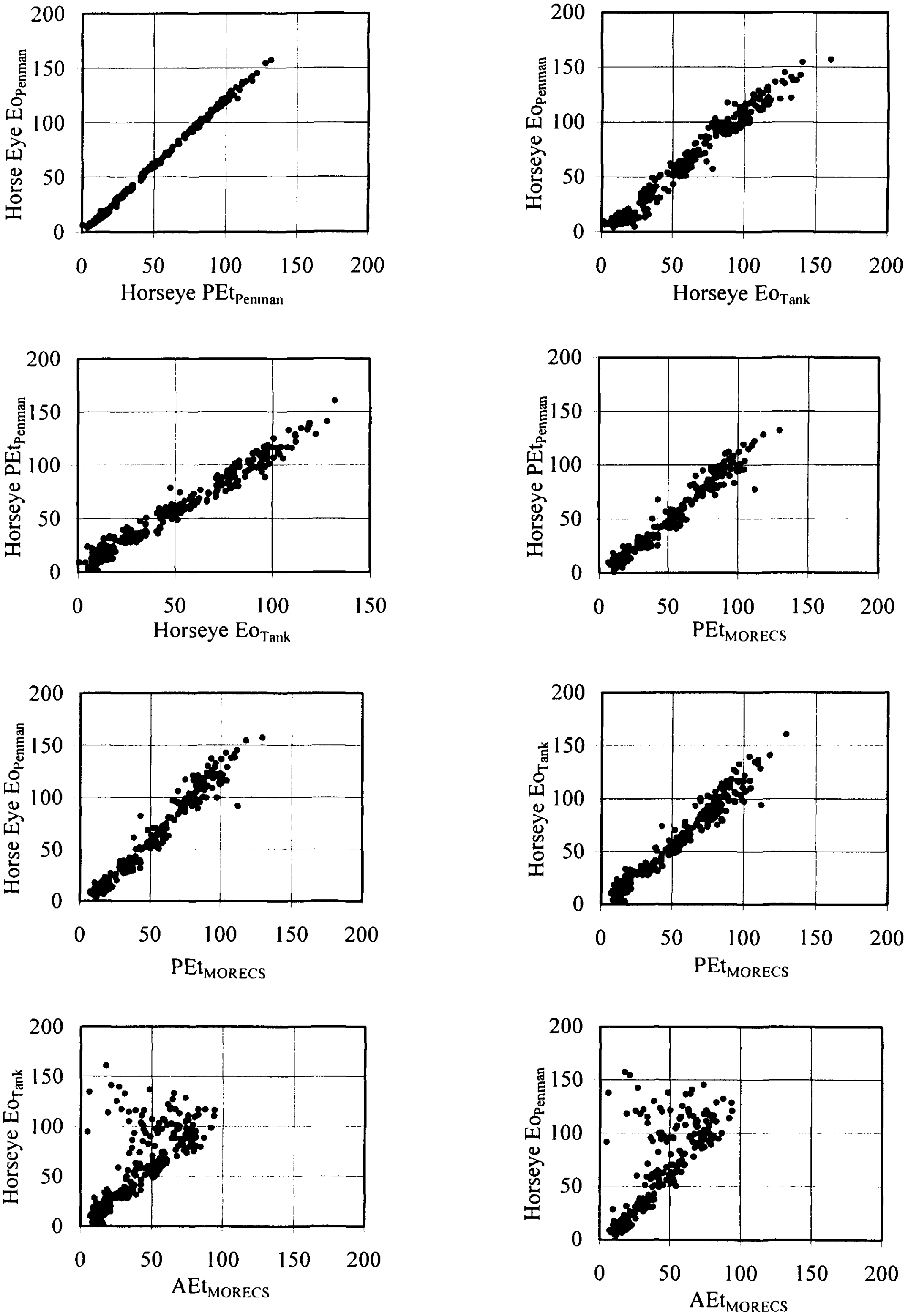


Figure 3.8. The relationships between the various evaporation and evapotranspiration estimates available on the Pevensey Levels wetland. Data shown have been employed to develop the regression matrix shown in Table 3.3.

3.4. The Hydrology of Embanked Channels

As stated in Chapter 2, the three main embanked channels on the Pevensey Levels (the Wallers Haven, Pevensey Haven and East Stream) play an important role in the catchment-scale hydrological functioning of the Pevensey Levels. This is because most pumping stations discharge into an embanked channel, with water level management in these channels dictating the proportion of pumped discharge that is lost to sea and the amount of water available for summer feeding. To date however, the hydrology of embanked channels on the Pevensey Levels has not been considered in detail, precluding quantitative assessments of losses to sea during winter and feeding of the lowland area during summer, as well as surface water storage within them.

The hydrological functioning of embanked channels is of even greater significance when considered in the context of revised water level management strategies on the Pevensey Levels. By assessing volumetric storage within the embanked channels, it is possible to determine the potential use of these channels as surface water reservoirs to provide water for the WES and WLMP targets. It may also be possible to establish the water level targets which need to be maintained in embanked channels at different times of year in order to supply water to ditches where revised water level management strategies may be in operation. Because losses to sea from the wetland are not measured directly, evaluations of the hydrology of embanked channels were considered based on a water balance approach applied individually to each of the three embanked channel systems. In conceptual terms, inputs to each of these embanked systems could be considered as precipitation falling directly on the water surface, surface inflows from streams and rivers and contributions from pumping stations discharging into each embanked channel system. Outflows were evaporation from the water surface, losses to sea and seepage. An important assumption was that the interaction between surface water and groundwater was negligible due to the low prevailing hydraulic conductivity of the soil (Section 2.3.). For each of these variables, this section describes the availability of data that can be employed to quantify them in embanked channel system. Essentially, these data are employed to estimate losses to sea from the wetland as well as water storage within the embanked channel system. In later sections, the latter data are employed to evaluate the potential role of embanked channels as reservoirs associated with the provision of water for revised water level management strategies instated on the wetland.

3.4.1. THE WALLERS HAVEN

The Wallers Haven is one of the most important components of the hydrology of the Pevensey Levels. The main surface water supply to the wetland is provided by the upland catchments of the Nunningham, Ashbourne, Hugletts and Ninfield streams (Douglas, 1993) that merge just north of Boreham Bridge to form the Wallers Haven. Figure 3.9 provides a detailed description of other features influencing the hydrology of the Wallers Haven. Data allowing the computation of all stores, inflows and outflows from the channel were available except data describing losses to sea (Q_{Sea}) during winter and lowland feeding during summer (Q_{feeding}). Therefore the sum of Q_{Sea} and Q_{feeding} could be quantified based on Figure 3.9 by inverting the water balance equation, so that

$$Q_{\mathrm{Sea}}+Q_{\mathrm{feeding}} = Q_{\mathrm{Boreham\ Bridge}}+ Q_{\mathrm{Star\ Inn\ PS}} + Q_{\mathrm{Horsebridge\ PS}} + R - E - A \pm \Delta S \quad \text{(Equation 3.11)}$$

where

$Q_{\mathrm{Boreham\ Bridge}}$	is flow at Boreham Bridge (m^3) ,
$Q_{\mathrm{Star\ Inn\ P.S.}}$	is discharge from the Star Inn pumping station (m^3),
$Q_{\mathrm{Horsebridge\ P.S.}}$	is discharge from the Horsebridge pumping station (m^3),
E	is evaporation from the water surface (m^3),
R	is rainfall falling on the water surface (m^3),
A	is abstraction for public water supply (m^3) and
ΔS	are changes in storage in the Wallers Haven (m^3).

A method to distinguish between losses to sea and the feeding of the lowland area has been developed, although an important limitation is the temporal resolution of available hydrological data. For example, pumping station discharge data required for the computation of $Q_{\mathrm{Star\ Inn\ PS}}$ and $Q_{\mathrm{Horsebridge\ PS}}$ are only collected on a weekly basis, and the days when data readings are made vary from station to station (Mick Phillips, EA Sluice Keeper, Pers. Comm.). This factor limited the development of a daily water balance for the calculation of Q_{Sea} and Q_{feeding}. Only on a monthly basis did all pumping station data coincide. Similarly, data describing surface water abstraction at Boreham Bridge, required to compute A in equation 3.11, were only available on a monthly basis. Consequently, a monthly time-step was chosen for the Wallers Haven water balance assessment, where necessary interpolating to derive daily or weekly values.

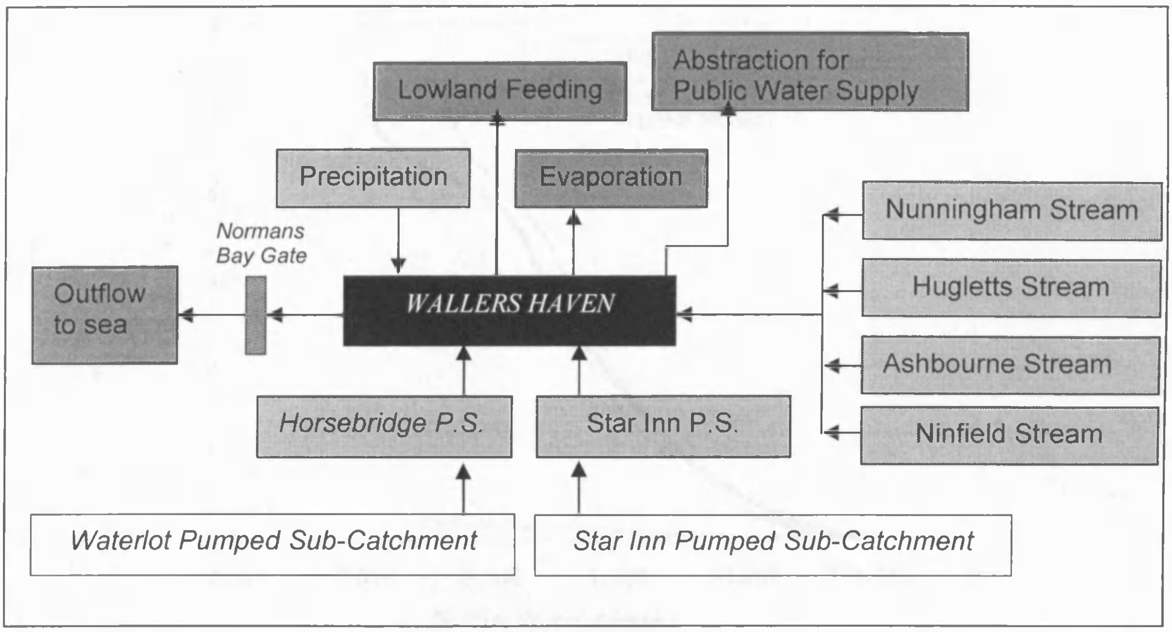


Figure 3.9. Conceptual representation of the water balance of the Wallers Haven and the eastern portion of the Pevensey Levels wetland.

For the calculation of $Q_{\text{Boreham Bridge}}$, data describing inflows from upland streams were applied within the factor formula described in Section 2.4.7. The accuracy of this method relative to direct measurement of flow is considered in Section 3.4.2. Potential implications regarding the suitability of the current surface water abstraction licence on the Wallers Haven are discussed in Section 3.7.3.

For all upland tributaries of the Wallers Haven except the Ninfield Stream, flow data were available since 1950. For the Ninfield Stream, data were only available since March 1985. Flow duration curves for the four upland streams (Figure 3.10) identified the largest flows associated with the Ashbourne Stream, and the smallest with the Ninfield Stream. For the upland streams, a strong seasonal component to flow was evident, especially relative to the proportion of rainfall generating runoff, termed the runoff coefficient (R_c). In terms of the proportions of rainfall generating runoff, the hydrological behaviour of the Hugletts and Nunnigham Streams was analogous. The highest runoff coefficients were recorded in the Ashbourne stream (Table 3.4), where a considerable proportion of the catchment area is woodland (Figure 2.5).

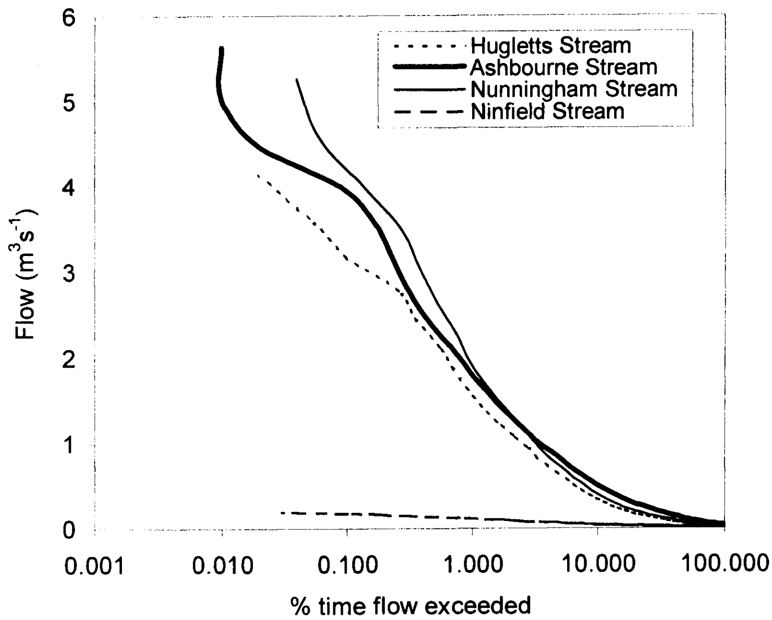


Figure 3.10. Flow duration curves for the upland tributaries of the Wallers Haven.

	Nunningham Stream		Ashbourne Stream		Hugletts Stream	
	Rainfall (mm)	Runoff (%)	Rainfall (mm)	Runoff (%)	Rainfall (mm)	Runoff (%)
Jan	85	79	88	78	84	74
Feb	57	82	56	91	56	80
Mar	59	63	63	70	61	61
Apr	52	44	54	63	52	48
May	51	24	51	43	50	28
Jun	56	14	58	26	57	18
Jul	57	11	61	18	58	12
Aug	68	9	68	16	68	10
Sep	74	11	78	18	76	12
Oct	92	22	96	30	93	24
Nov	96	46	101	48	97	44
Dec	94	61	94	59	90	58

Table 3.4. Mean monthly rainfall and runoff for tributaries of the Wallers Haven, calculated from data 1950-1996.

In the Wallers Haven catchment, contributions from upland streams are complemented by discharge from two pumping stations. The total catchment area of the Wallers Haven can therefore be considered as the combination of the catchments of its four tributaries as well as the drainage from the Star Inn and Horsebridge pumping stations. The total area of these combined catchments is 105.3km^2, illustrating the importance of the Wallers Haven with regards to the overall hydrological functioning of the wetland. For the water balance assessment, inflows from the Star Inn and Horsebridge pumping stations were calculated based on the method proposed by Marshall (1989). Since the pumping stations have stated fixed capacities, the volume of water discharged by a pumping station on a monthly basis (Q_{Pump}) can be calculated by

$$Q_{\mathrm{Pump}} = \text{Pump Capacity} \times [3600 \times \text{Hours Pumped}] \qquad \text{(Equation 3.12)}$$

where pump capacity is in m^{3}s^{-1}. For all pumping stations on the wetland, the capacity of individual pumps has been previously given in Table 2.7. However, this calculation has to be conducted for individual pumps at each pumping station since different pump combinations operate at different times of year (Section 2.4.3).

Changes in storage, and the contributions of evaporation and precipitation to the Wallers Haven were estimated using level-volume-area relationships developed from descriptions of the geometry and dimensions of the channel. A level-volume-area relationship (LVAR) is an empirical model relating water level to volumetric storage and the water surface area expressed as regression algorithms (Reed, 1993). The approach has been widely applied in wetland hydrological studies (Sutcliffe and Parks, 1977; Thompson and Hollis, 1995; Thompson and Hollis, 1998) and is employed extensively in later chapters to model the hydrology of field scale ditches on the wetland. For water level l, channel storage (S_l)(m^3) can be calculated by

$$S_l = CSA_{\mathrm{Channel}\ l} \times L_{\mathrm{Channel}} \qquad \text{(Equation 3.13)}$$

where $CSA_{\mathrm{Channel}\ l}$ is the cross-sectional area of the channel at water level l (m^2), and L_{Channel} is the length of channel represented by the cross-section (m). For the equivalent water level, the surface water area ($A_{\mathrm{Surface\ Water}\ l}$)(m^2) can be obtained from

$$A_{\mathrm{Surface\ Water}\ l} = Ww_{\mathrm{Channel}\ l} \times L_{\mathrm{Channel}} \qquad \text{(Equation 3.14)}$$

where $Ww_{Channel\ l}$ is the cross-section water width at level l (m). $CSA_{Channel\ l}$, $L_{Channel}$ and $Ww_{Channel\ l}$ could all be quantified from survey data available at the EA Pevensey office. 10 channel cross-sections were digitised from EA records and are shown in Figure 3.11. Based on the total length of the Wallers Haven (9.4 km), each cross section was representative of a stretch of channel approximately 1km in length. CSAs did not vary greatly along the longitudinal profile of the channel (maximum 82 m^2, minimum 55 m^2). Actual cross sectional dimensions were wider and shallower than those proposed by Newbold *et al.* (1989) for Type 4 channels in wet grassland areas. However, actual CSAs at bankfull were, on average 84 % of the CSAs suggested by Newbold *et al.* (1989), supporting the use of this classification scheme for the development of LVARs for embanked channels where no cross-sectional data are available.

The level-volume and level-surface area relationships for the Wallers Haven are shown in Figure 3.12. These relationships have been obtained by regression of channel volumetric storage / open water area relative to water levels in the Wallers Haven at Norman's Bay ranging from the bankfull to the bed level. Water levels in the Wallers Haven are measured from a stageboard at Norman's Bay on a daily basis, so that the time-series available is highly suitable for the estimation of surface water storage, and the contributions and losses from the channel surface by direct precipitation and evaporation respectively. Horseye Eo_{Tank} data were employed to represent evaporation from the water surface. Rainfall data from Hazard's Green, the closest raingauge to Boreham Bridge, was used to represent direct channel precipitation.

In combination, all these data allowed the resolution of equation 3.11 and provided an indication of the magnitude of Q_{Sea} and $Q_{feeding}$. However, for the purpose of the water balance assessment it was necessary to differentiate between the two. There are over 30 sluices along the Wallers Haven that can be employed to feed the lowland area, but few data on their operation were available (M. Phillips, EA Sluice Keeper, Pers. Comm.). Data regarding the management of the Norman's Bay Gate were therefore employed to distinguish between the two processes. When water levels in the Wallers Haven at the start of the time-step were below the Norman's Bay gate retention level, generally 2.89 m O.D. in summer and 1.64 m O.D. in the winter (M. Phillips, EA Sluice Keeper, Pers. Comm.), any losses from the channel were ascribed to wetland 'feeding'. In contrast, where initial water levels were above the retention level, any losses were ascribed to Q_{Sea} assuming that losses by evaporation were negligible.

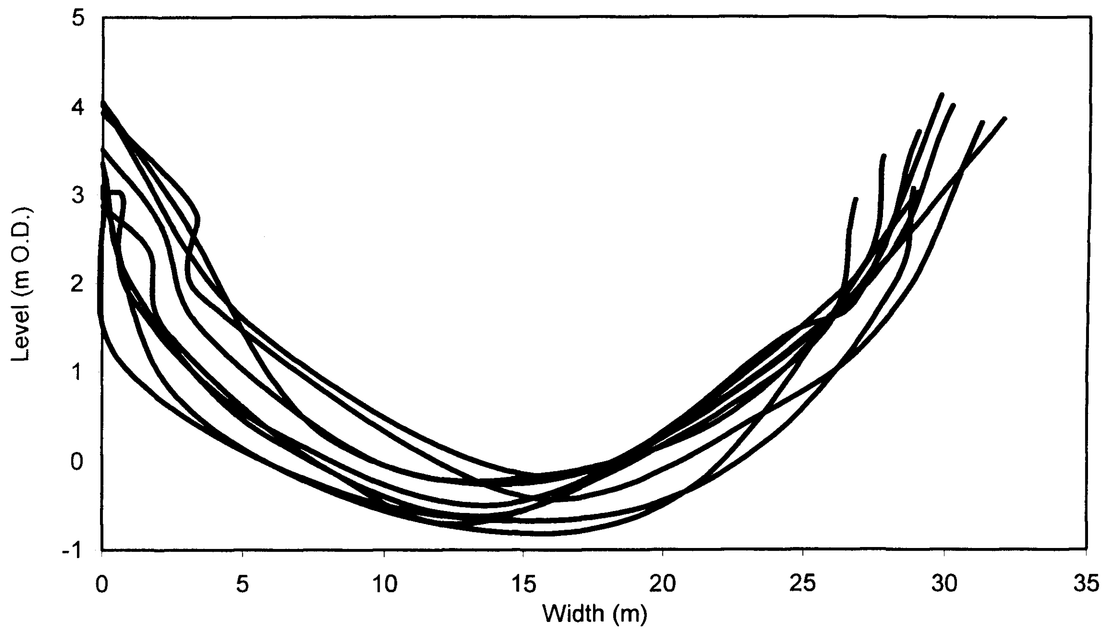


Figure 3.11. Cross sections of the Wallers Haven (from Environment Agency records) employed in the calculation of the level-volume-area relationships.

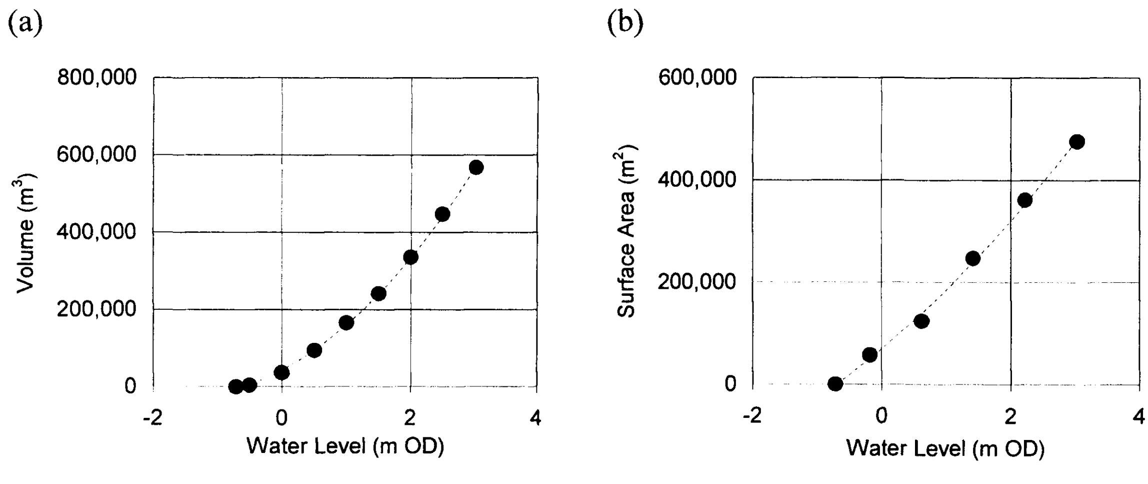


Figure 3.12. (a) Level-Volume and (b) Level-Area relationships for the Wallers Haven. Regression algorithms for calculation are shown in Table 3.6.

A component water balance for the Wallers Haven for the period between 1995 and 1998 is shown in Figure 3.13. Results corroborated the seasonal aspects of water level management in embanked channels, where storage in the summer and discharge in the winter are the key objectives (Section 2.4.4). During the winter months most inflows by-passed the wetland and close to 100 % of the inflows to the Wallers Haven were discharged out to sea (Figure 3.14.a). During the summer, losses to sea were limited and a smaller proportion of inflows are employed to feed the lowland area (Figure 3.14.a). Figure 3.13 shows that direct rainfall and evaporation are minor components of the water balance and that the volumes of water employed for feeding can exceed abstraction for public water supply (Figure 3.13). This result provides an indication of the likely effects of abstraction on the wetland, concerns regarding which have been previously raised by local stakeholders, especially conservationists. The proportion of total monthly inflows abstracted for the period between 1995 and 1998 amounted to 34 %. In all summers during the study period, the proportion of total inflows abstracted from the Wallers Haven exceeded 50% during at least one month. In the drier summers of 1995 and 1996, abstraction exceeded 50% of all inflows on five and six months respectively, although augmentation from the groundwater boreholes was a feature of all summers included in the analysis.

The need for augmentation to support the current abstraction licence is clearly illustrated in Figure 3.14.b. In that figure, abstraction as a proportion of 'naturalised' flows, defined as the volume of water supplied by the four upland tributaries of the Wallers Haven excluding the augmentation volume, is compared to the proportion of actual flows abstracted. Based on this analysis, the proportions of inflows abstracted from the Wallers Haven represented 45% of naturalised inflows over the entire four year period. In 1995 and 1996, over 100% of naturalised inflows were abstracted (Figure 3.14.b), suggesting that the water company would have difficulties in satisfying demand in dry years without the availability of mitigating measures such as augmentation. Other aspects of the required flexibility of water level managament on the wetland were also apparent in the data. The overall objectives of water storage in the summer and drainage in the winter are illustrated by the volumes of water lost to sea in different years. During the wetter years of 1997 and 1998, losses to sea were more than twice those calculated for 1995 and 1996. During dry summers, a large proportion of pumped inflows were re-directed to the lowland network (Figure 3.14.c), highlighting potential in-efficiencies in summer pump-functioning.

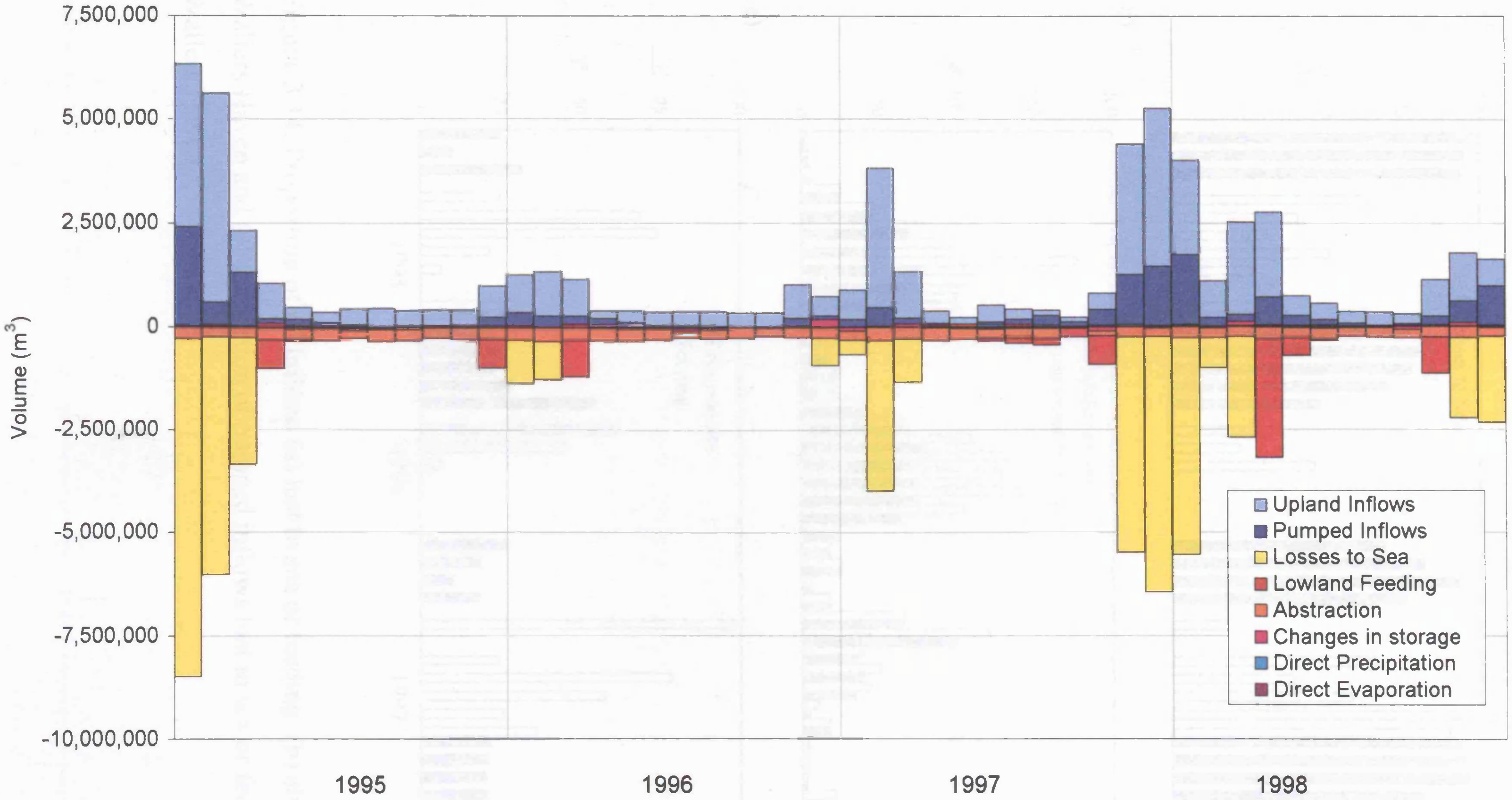


Figure 3.13. Monthly component water balance for the Wallers Haven between 1995 and 1998.

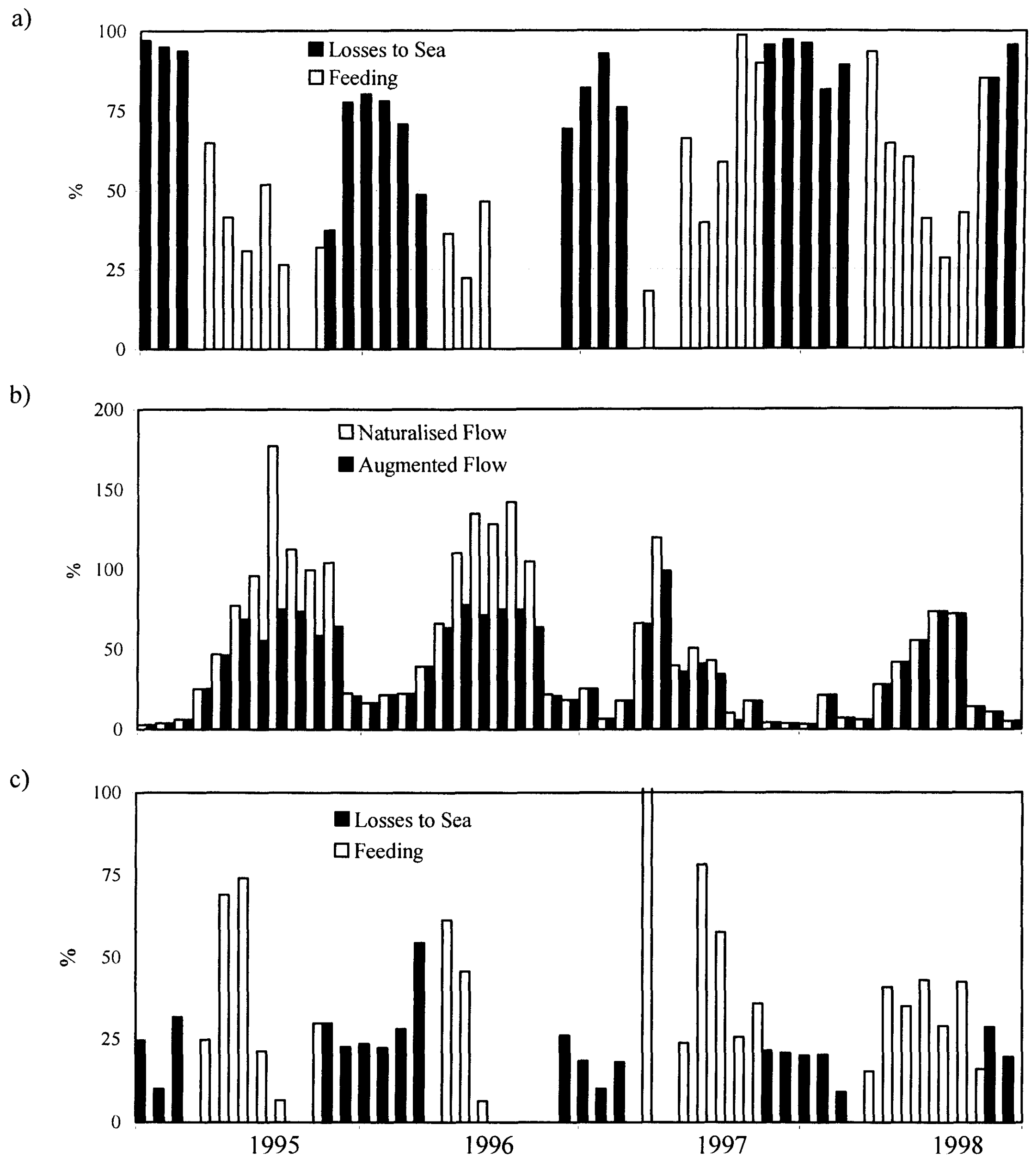


Figure 3.14. Proportion of all inflows (a) lost to sea or feeding, (b) abstracted from the Wallers Haven and (c) proportion of pumped inflows lost to sea or feeding from the Wallers Haven

3.4.2. ACCURACY OF FLOW ESTIMATES AT BOREHAM BRIDGE

Previous work by Douglas and Hart (1993) has suggested that the factor formula currently employed for the estimation of flow in the Wallers Haven, described in section 2.4.7, over-estimates actual flows during low flow conditions and endangers the attainment of the MRF. In recent years, instrumentation capable of addressing these concerns has been installed at Boreham Bridge. An ultrasonic flowmeter has been located at Boreham Bridge since 1995. However, due to continuing problems with vandalism (Section 2.4.7), only 133 days of data were available for the period covered by the wetland water balance. Although their duration was undoubtedly insufficient to provide a detailed assessment of the performance of the factor formula, data provided by the ultrasonic gauge were employed to evaluate the accuracy of existing flow estimates, thus evaluating potential errors in water balance calculations.

Comparison between ultrasonic and factor formula derived flow data suggested that both data sets were of good quality and comparable, as illustrated by the generally constant slope provided by double mass analysis between the two flow estimates (Figure 3.15). For the period for which data was available, time-series of flow measured by the ultrasonic gauge and estimated by the factor formula are shown in Figure 3.16. To evaluate a potential flow dependency of the perceived inaccuracies of the factor formula, comparisons between the flow data provided by the two estimates were segregated according to high or low flows. High flows were defined as those estimated by the factor formula that were in excess of 0.2 m^3s^{-1}. Although on a daily basis between 1970 and 1998, factor formula flows at Boreham Bridge exceeded 0.2 m^3s^{-1} for 51% of the time, for the period for which ultrasonic data were available, only 14 days, 10.5% of the total data period, were characterised by high flows. For the majority of the period for which data were available, low flows, defined as those estimated by the factor formula less than 0.2 m^3s^{-1}, dominated the hydrological regime of the Wallers Haven.

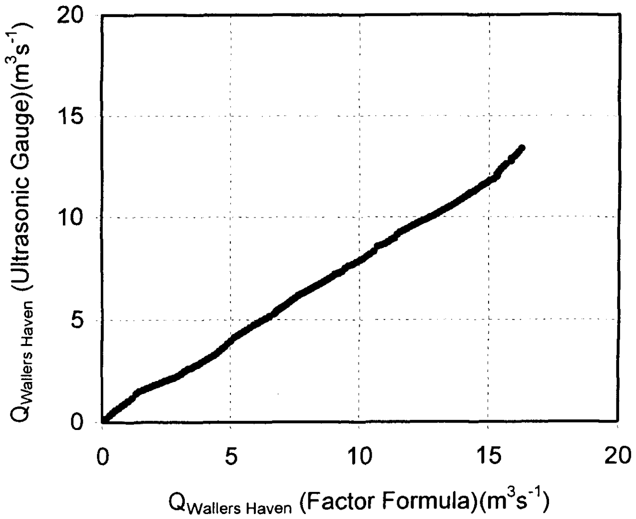
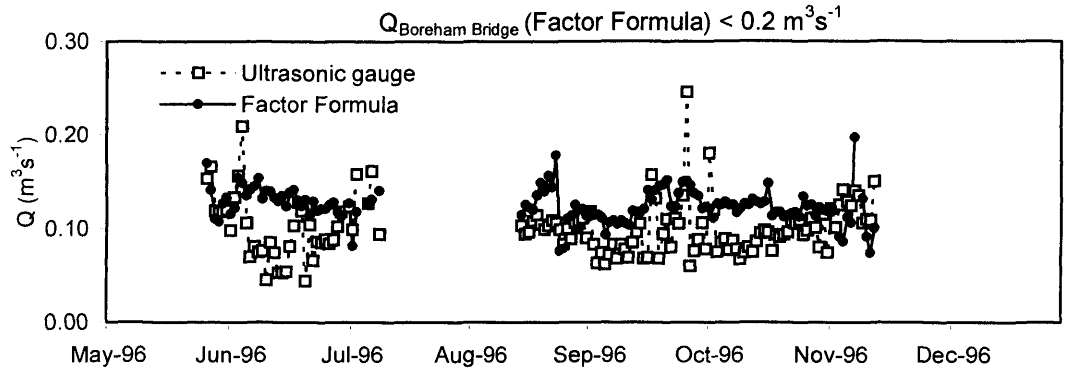


Figure 3.15. Double mass analysis of Boreham Bridge flow estimates (in cumecs).

(a)



(b)

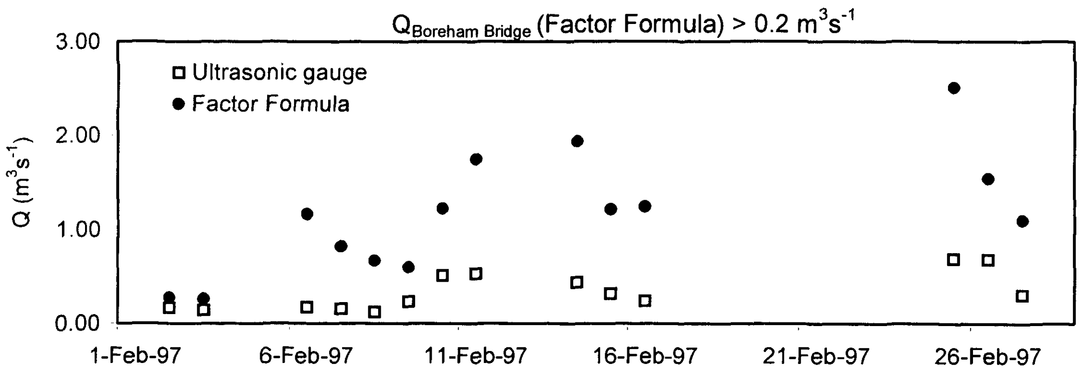


Figure 3.16. Comparison between estimates of flow in the Wallers Haven at Boreham Bridge for the second half of 1996. Factor formula data shown are limited to periods when ultrasonic flow gauge data are available.

Comparison between flow estimates supported suggestions by Hart and Douglas (1993) regarding the inadequacy of the factor formula method for estimating flow at Boreham Bridge. For the 133 days for which ultrasonic flow data were available, flow calculated using the factor formula exceeded ultrasonic estimates for 82 % of the time. On average, the factor formula over-estimated flows by 0.10 m^3s^{-1}, equivalent to 8640 m^3day^{-1} or 50% of the daily abstraction licence (see Section 2.4.7). However, this precluded the apparent flow dependence of the relationships. The highest differences were recorded at flows in excess of 0.2 m^3s^{-1}. At these flows, the mean difference between ultrasonic and factor formula flow data was 0.83 m^3s^{-1}. At flows less than 0.2 m^3s^{-1}, smaller variations of 0.02 m^3s^{-1} equivalent to 1728 m^3day^{-1} or 10% of the daily abstraction licence were apparent.

During low flow conditions, the relationship between ultrasonic and factor formula flow estimates was characterised by an extensive degree of scatter, a feature of the data highlighted by the low coefficient of determination obtained by the regression of the two estimates (Figure 3.17.a). In contrast, for flows in excess of 0.2 m^3s^{-1} the two estimates were more closely related (Figure 3.17.b). At high flows, the most appropriate means of adjusting historical flow data was based on the relationship between factor formula flow ($Q_{\text{Boreham Bridge FF}}$) and the difference between flow estimated by factor formula and ultrasonic gauge methods ($Q_{\text{Boreham Bridge Difference}}$). Results shown in Figure 3.18.b indicate that, for flows in excess of 0.2 m^3s^{-1}, the flow at Boreham Bridge ($Q_{\text{Boreham Bridge}}$ Actual) could be calculated by

$$Q_{\text{Boreham Bridge (Actual)}} = Q_{\text{Boreham Bridge FF}} - 0.72Q_{\text{Boreham Bridge FF}} \qquad \text{(Equation 3.15)}$$

where $Q_{\text{Boreham Bridge FF}}$ is in m^3s^{-1}. In contrast, the low coefficient of determination associated with the equivalent relationship for low flow conditions (Figure 3.18.a) highlighted the difficulty of accurately estimating the actual volumetric contributions of the Wallers Haven to the wetland during dry summers. No significant trends were apparent in the relationship between $Q_{\text{Boreham Bridge FF}}$ and $Q_{\text{Boreham Bridge Difference}}$, an important result in the context of abstraction. Difficulties in estimating Wallers Haven flow coincided with the crucial summer months when flows were at an annual minimum but abstraction demand was highest. During the summer of 1996 (May-September), although flows were less than 0.2 m^3s^{-1} for 98.8% of the entire period, 1.42 million m^3 were abstracted from the Wallers Haven.

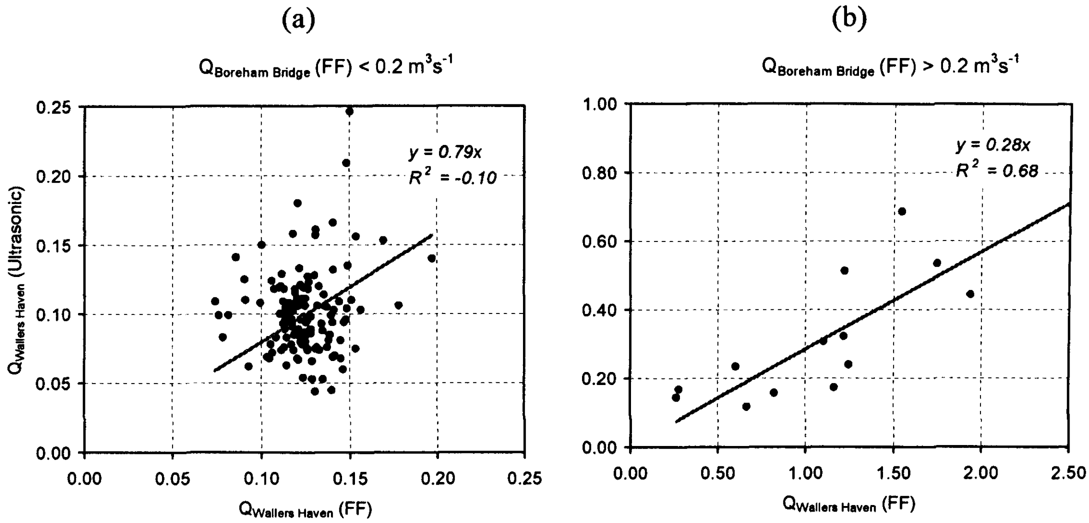


Figure 3.17. Relationship between flows estimated at Boreham Bridge using the factor formula ($Q_{\text{Boreham Bridge FF}}$) and the ultrasonic flow gauge ($Q_{\text{Boreham Bridge Ultrasonic}}$). All data in cumecs.

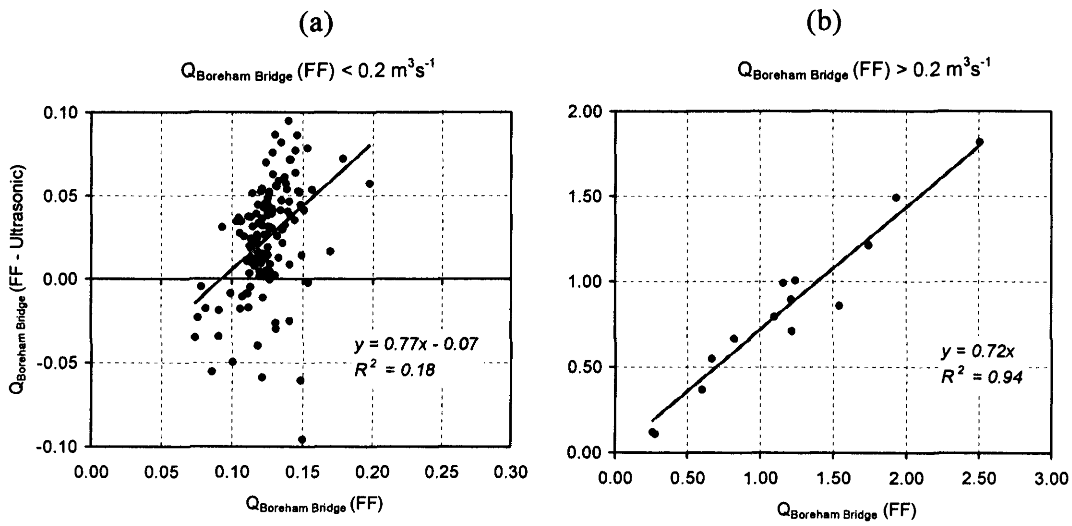


Figure 3.18. Comparison between flow in the Wallers Haven, estimated by the factor formula method, relative to the difference between the factor formula method and the ultrasonic gauge. All data in cumecs.

3.4.3. THE PEVENSEY HAVEN AND EAST STREAM

3.4.3.1. Channel dimensions and level-volume relationships

The calculation of losses to sea from the Pevensey Haven and East Stream required the application of a method analogous to that employed for the Wallers Haven. Level-volume relationships were developed from the cross-sectional and longitudinal dimensions of the channels provided by the Environment Agency. The cross-sectional data employed are shown in Figure 3.19. Level-volume relationships for each of the embanked channels on the wetland are shown in Table 3.5. The mean cross-sectional dimensions of the East Stream, Glynleigh Haven and Hurst Haven were most closely associated with Type 2 channels in the classification proposed by Newbold *et al.* (1989). These channels had cross-sectional dimensions 96%, 122% and 137% respectively of areas calculated for Type 2 channels, although in common with results obtained for the Wallers Haven, channels tended to be wider and shallower than those proposed by Newbold *et al.* (1989). The cross-sectional dimensions of the Chilley Stream were most accurately represented by Type 3 ditches, and on average, cross-sectional dimensions of that channel were 75% of their dimensions. The cross-sectional dimensions of the Pevensey Haven were similar to the Wallers Haven, although this result could not be fully verified as only one sample cross-section was available for the Pevensey Haven (Figure 3.19). The mean dimensions of the Pevensey Haven were 85% of Type 4 ditches compared to 98% for the Wallers Haven cross-sections.

For the East Stream and Hurst Haven channel systems, there was some evidence for the observation of tapering drains by Reed (1985) (Section 1.6.1). For all embanked channels on the wetland, bed levels relative to the distance from the retention gate, termed chainage, are shown in Figure 3.20.a. With increasing chainage, bed levels in all embanked channels tended to increase, especially in the case of the East Stream and Hurst Haven. For these channels, cross-sectional dimensions decreased with upstream distance (Figure 3.20.b). However, these trends were not apparent for channels other than the East stream and Hurst Haven, and the relationship between chainage and both cross-sectional dimensions and bed level was characterised by a greater degree of scatter. Results suggested caution when applying known models of drainage network design on the Pevensey Levels wetland, especially given the difficulty of placing all embanked channels on the Pevensey Levels wetland within one of the channel classes proposed by Newbold *et al.* (1989).

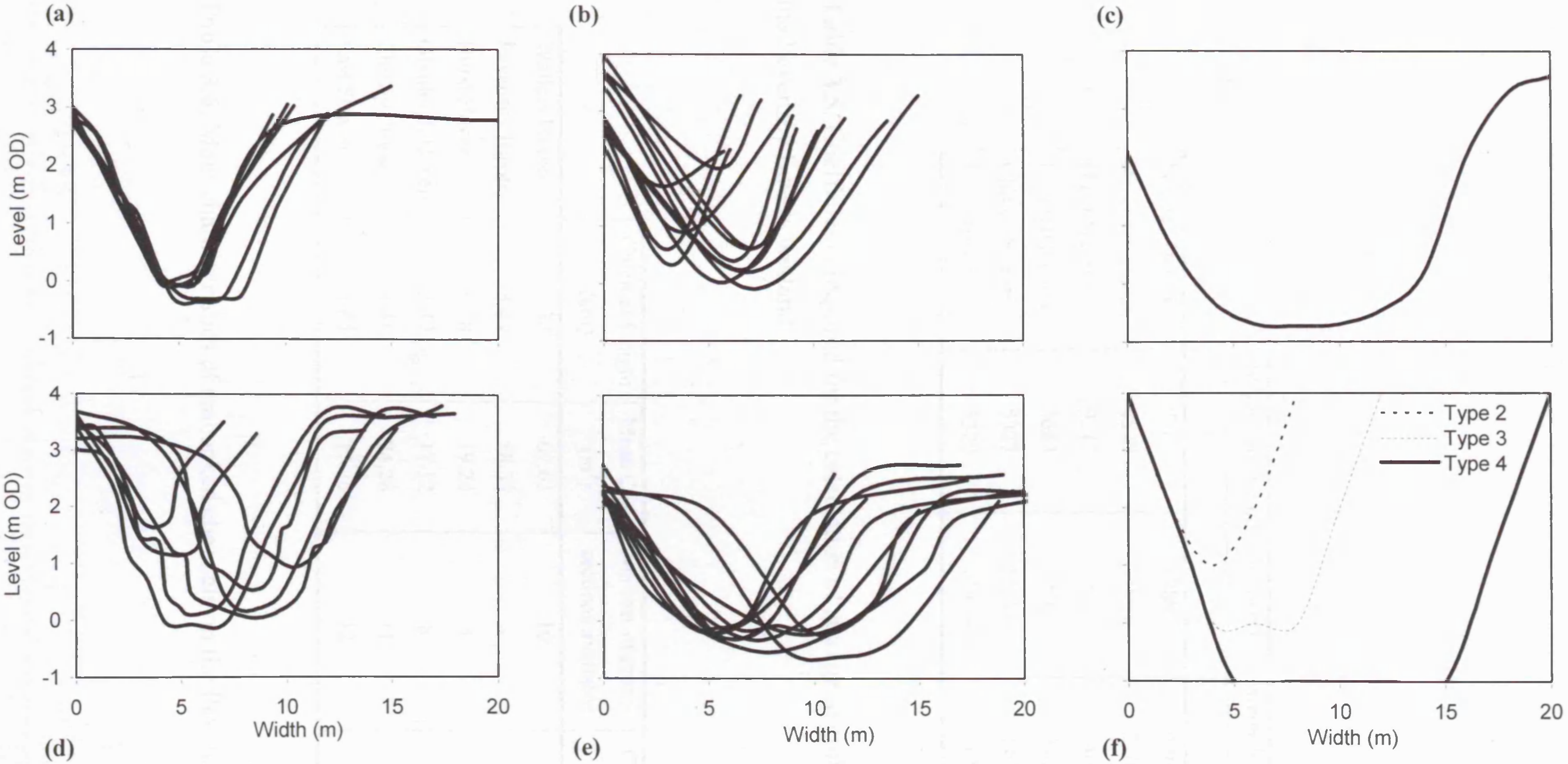


Figure 3.19. Cross-sectional data employed to develop level-volume-area relationships for embanked channels: (a)Glynleigh Haven, (b) East Stream, (c) Pevensey Haven, (d) Hurst Haven, (e) Chilley Stream and (f) the typical dimensions proposed by Newbold *et al.* (1989).

	Storage (m^3) = a(water level)^2 + b(water level) +c		
	a	b	c
Wallers Haven	27827	89897	43606
Pevensey Haven	4829	35961	22047
Hurst Haven	9342	5526	404
Glynleigh Haven	3663	7506	1855
Chilley Stream	5327	15337	5612
East Stream	3320	12650	125

Table 3.5. Coefficients required for the calculation of storage in embanked channels of the Pevensey Levels wetland.

	Channel Length (km)	Mean CSA (m^2)	Number of cross-sections available	CSA (% of Newbold *et al.*, 1989)
Wallers Haven	9.32	67.63	10	98% of Type 4
Pevensey Haven	3.49	58.53	1	85% of Type 4
Hurst Haven	6.28	19.24	8	137% of Type 2
Glynleigh Haven	3.49	17.12	8	122% of Type 2
Chilley Stream	3.41	24.26	11	75% of Type 3
East Stream	4.85	13.48	12	96% of Type 2

Table 3.6. Main characteristics of embanked channels on the Pevensey levels wetland.

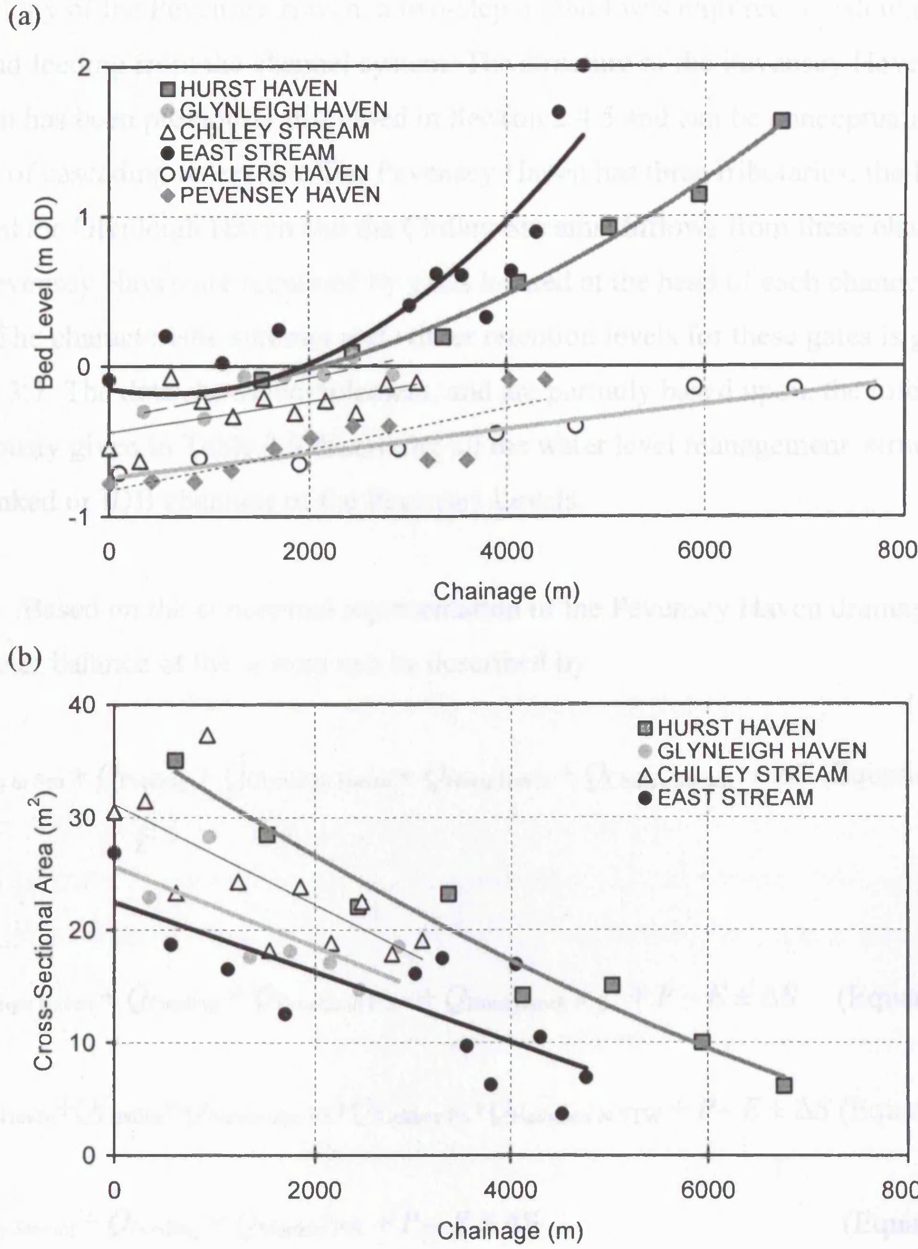


Figure 3.20. Relationship between (a) bed levels and upstream distance (chainage) and (b) cross-sectional dimensions and upstream distance (chainage) for embanked channels on the Pevensey Levels wetland (data for the Wallers Haven are not shown. For the Pevensey Haven only one cross-section was available (Source of data: Environment Agency).

3.4.3.2. Losses to sea

Losses to sea and feeding from the Pevensey Haven were quantified by application analogous to that employed for the Wallers Haven. However, due to the complex hydrology of the Pevensey Haven, a two-step method was required to calculate losses to sea and feeding from the channel system. The structure of the Pevensey Haven drainage system has been previously discussed in Section 2.4.5 and can be conceptualised as a series of cascading reservoirs. The Pevensey Haven has three tributaries, the Hurst Haven, the Glynleigh Haven and the Chilley Streams. Inflows from these channels into the Pevensey Haven are regulated by gates located at the head of each channel (Figure 2.7). The characteristic summer and winter retention levels for these gates is given in Table 3.7. The data shown complement, and are partially based upon, the information previously given in Table 2.9 describing all the water level management structures on embanked or IDB channels of the Pevensey Levels.

Based on the conceptual representation of the Pevensey Haven drainage system, the water balance of the system can be described by

$$Q_{\text{Losses to Sea}} + Q_{\text{Feeding}} = Q_{\text{Glynleigh Haven}} + Q_{\text{Hurst Haven}} + Q_{\text{Chilley Stream}} \pm \Delta S \quad \text{(Equation 3.16)}$$

where

$$Q_{\text{Glynleigh Haven}} + Q_{\text{Feeding}} = Q_{\text{Drockmill P.S.}} + Q_{\text{Honeycrock P. S.}} + P - E \pm \Delta S \quad \text{(Equation 3.17)}$$

$$Q_{\text{Hurst Haven}} + Q_{\text{Feeding}} = Q_{\text{Newbridge PS}} + Q_{\text{Rickney PS}} + Q_{\text{Hailsham N STW}} + P - E \pm \Delta S \quad \text{(Equation 3.18)}$$

$$Q_{\text{Chilley Stream}} + Q_{\text{Feeding}} = Q_{\text{Manxey P.S.}} + P - E \pm \Delta \text{S} \quad \text{(Equation 3.19)}$$

where in each case the subscript denotes the source of the inflow. The main inflows to most of the tributary channels are from pumping stations located along their length, although no pumping station discharges directly into the Pevensey Haven. Pumped inflows for each of the tributary channels were calculated based on the method advocated by Marshall (1989) (Equation 3.12) based on monthly pump hour data for each station between 1995 and 1998 provided by the Environment Agency. Other inflows to tributary channels are from minor streams. However, few data describing their hydrological regime were available, so that analyses were conducted assuming the negligible influence of these streams in terms of the wetland water balance, as proposed by Douglas (1993) (Section 2.4.4). As a result, minor stream inflows were not considered in the water balance calculations. An important contribution to the Hurst Haven system is the Hailsham North sewage treatment works. The location of these works has been previously shown in Figure 2.4. Inflows from the remaining sewage treatment works at Lunsford's Cross, Hooe, Herstmonceux and Hailsham South are considered in later sections, since none of these discharge into embanked channels but are instead associated with the hydrology of individual pumped drainage units. For all sewage treatment works, discharge data were obtained from Southern Water.

As in the case of the Wallers Haven, for each of the tributary channels of the Pevensey Haven, water level data coupled to data describing the annual operation of retention gates were employed to distinguish between outflows from the channel system and episodes of lowland feeding. An equivalent approach was employed to distinguish losses to sea from feeding for the Pevensey Haven. For all gates associated with the Pevensey Haven system, winter and summer retention levels are given in Table 3.7.

A simpler approach was employed to calculate losses to sea from the East Stream system. In conceptual terms, the structure of the East Stream drainage system could be considered analogous to that of the Wallers Haven, since a series of upland streams (Figure 2.5, Figure 2.7) feed the top end of the system. However, as with minor streams feeding the Pevensey Haven system, few data were available describing their hydrological regime so that losses to sea from the system were calculated by

$$Q_{\text{Losses to Sea}} = Q_{\text{Barnhorn P.S.}} + P - E \pm \Delta S \qquad \text{(Equation 3.20)}$$

Total losses to sea from the Pevensey Levels wetland on a monthly basis for the period between 1995 and 1998 were then calculated by combining estimates for the Wallers Haven, Pevensey Haven and East Stream systems. For the three embanked channel systems on the wetland, losses to sea and lowland feeding are shown in Figure 3.21. Results are discussed in later sections in the context of wetland water availability and a number of proposed water level management strategies.

Gate	ID	Summer Level (m OD)	Winter Level (m OD)	NOTES
Pevensey Bridge	P 33	1.10	0.30	Minimum –0.2
Rickney Bridge	G 23	1.20	0.30*	
Rickney Automatic	R 03	1.89	1.50	Minimum 0.5
Chilley Bridge	P 07	1.50	0.30*	
Norman's Bay	S 36	2.89	1.64	Winter level = 2.0m when dry
East Stream Railway	S 10	1.40	0.30	

Table 3.7. Typical summer and winter retention levels for embanked channels gates. Gates at Chilley and Rickney Bridge become redundant in winter and water levels are controlled by the gate at Pevensey Bridge (Mick Phillipps, Pers. Comm.).

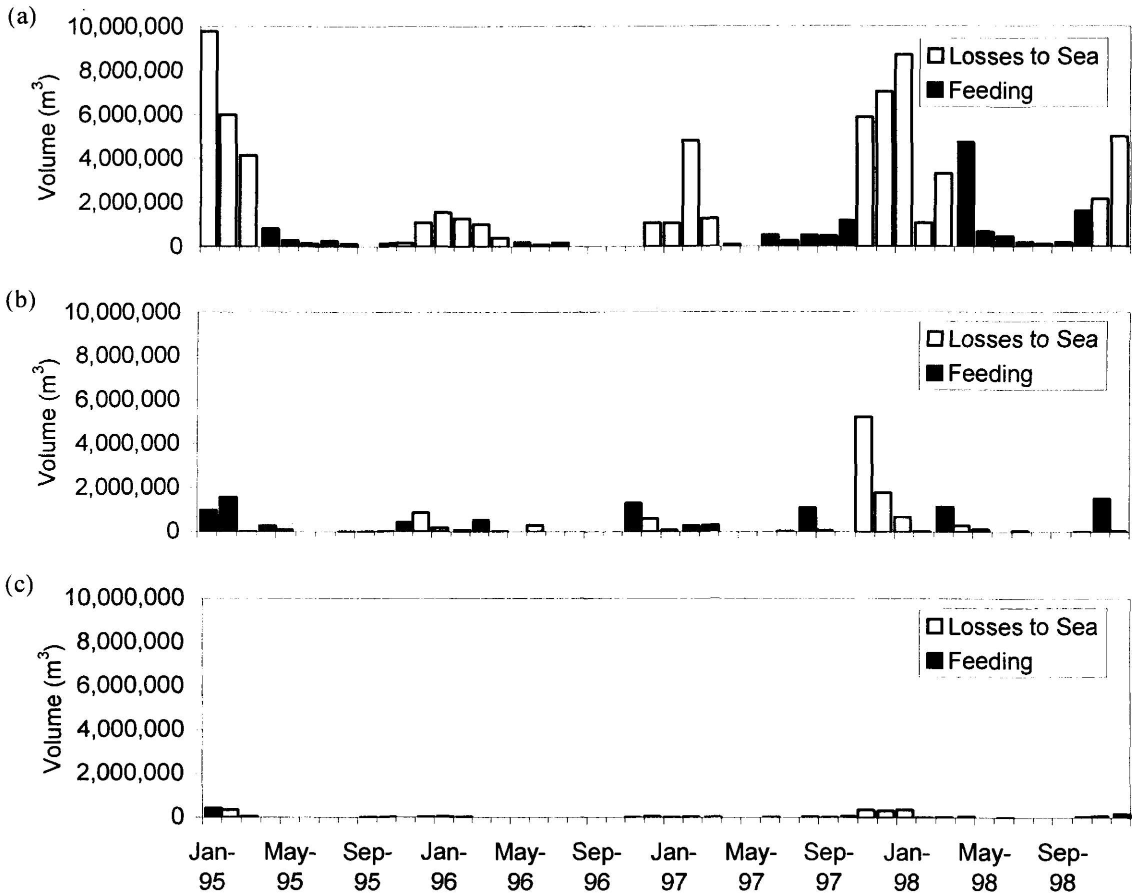


Figure 3.21. Volumes of water discharged to sea or employed for lowland feeding on a monthly basis from the (a) Wallers Haven, (b) Pevensey Haven and (c) East Stream embanked channel systems between 1995 and 1998. Volumes are shown on the same y-axis scale for comparative purposes.

3.5. Hydrology of Pumped Sub-catchments

In common with other aspects of the hydrology of the Pevensey Levels wetland, although data describing pump functioning are routinely collected, the dynamics of these systems had not been previously considered. In the context of the catchment water balance, the study of pump functioning is significant for a number of reasons. Firstly, as identified in Figure 3.21, pumping stations play an important role in determining losses to sea from the wetland. An understanding of the processes governing the hydrological functioning of the pumped sub-catchments is therefore essential if a predictive method for the calculation of losses to sea is to be a realistic target. Secondly, given the extensive lengths of ditches contained within pumped areas, surface water storage in the pumped sub-catchment channel network is likely to be considerable. The quantification of ditch storage in the lowland channel network is of particular significance in the context of the revised water level management strategies that are commonly instated as part of many wetland restoration schemes. Many of the areas identified as targets for restoration in WLMPs (Section 2.8.2) are within pump drained areas. The volumes of water currently stored in the lowland channel network can be compared against the volumes of water stored under various restoration scenarios. When considered in the context of the catchment water balance, this analysis can be employed to evaluate the sustainability of proposed increases in ditch water levels in water resource terms.

The main objective of this section is therefore to quantify the volume of water stored within pumped sub-catchments on the wetland based on the water level management *status quo*. In doing so, it provides a method for the estimation of ditch water storage under different water level management scenarios, including those associated with restoration strategies in wet grassland areas. It also considers the hydrological dynamics of pump-drained areas, examining the potential for the development of predictive tools for the estimation of the volumes of water pumped from the wetland, and providing a more spatially distributed approach to the assessment of wetland hydrological functioning than discussed in previous sections. Such issues of spatial scale are examined further in Section 3.6, which considers the hydrology of the Pevensey Levels wetland at the field scale.

3.5.1. WATER LEVEL VARIATIONS

The method employed for the calculation of storage in the ditches of pumped sub-catchments on the Pevensey Levels wetland was analogous to that employed to assess storage in embanked channels, and required the development of level-volume relationships for each pumped sub-catchment on the wetland. The calculation of surface water storage and the investigation of the dynamics of pump-drained areas on the wetland was possible due to the large volume of data available describing the hydrological functioning of each pump-drained system. Data describing the water levels in pumped drains and hours pumped at each pumping station on the wetland are routinely collected by the EA. These data were available for the period between 1995 and 1998 in the case of water level data, and between 1984 and 1998 for pump-hour data. However, one of the primary limitations of pumping station data was that the days when each pumping stations were visited were different. Although data are collected on a roughly weekly basis, for any given day, data for all pumping stations on the wetland were not necessarily available, and only on the first day of every month were wetland-wide pumping data coincident. As a result, the calculation of storage in pumped sub-catchments was limited to a monthly time-step. This time-step is equivalent to the temporal resolution of methods employed to quantify storage in the embanked channel systems on the Pevensey Levels (Section 3.3).

For pumping stations associated with the three main embanked channel systems on the wetland, weekly observations of water levels in pumped drains between January 1995 and December 1998 are shown in Figure 3.22. Distinctive seasonal trends were apparent within the water level records. Summer water levels were generally higher than those promoted in winter, in accordance with agricultural objectives (Section 1.6.3). Mean monthly water levels for the years between 1995 and 1998 for each pumping station are shown in Figure 3.23. In general, pumping stations maintained water levels equivalent to electrode settings at individual pumping stations (Table 2.7). Exceptions were during the winters of 1994-95 and 1997-98. However, the temporal resolution the data afforded did not allow the visualisation of the large variations in water levels that are known to occur during individual storms.

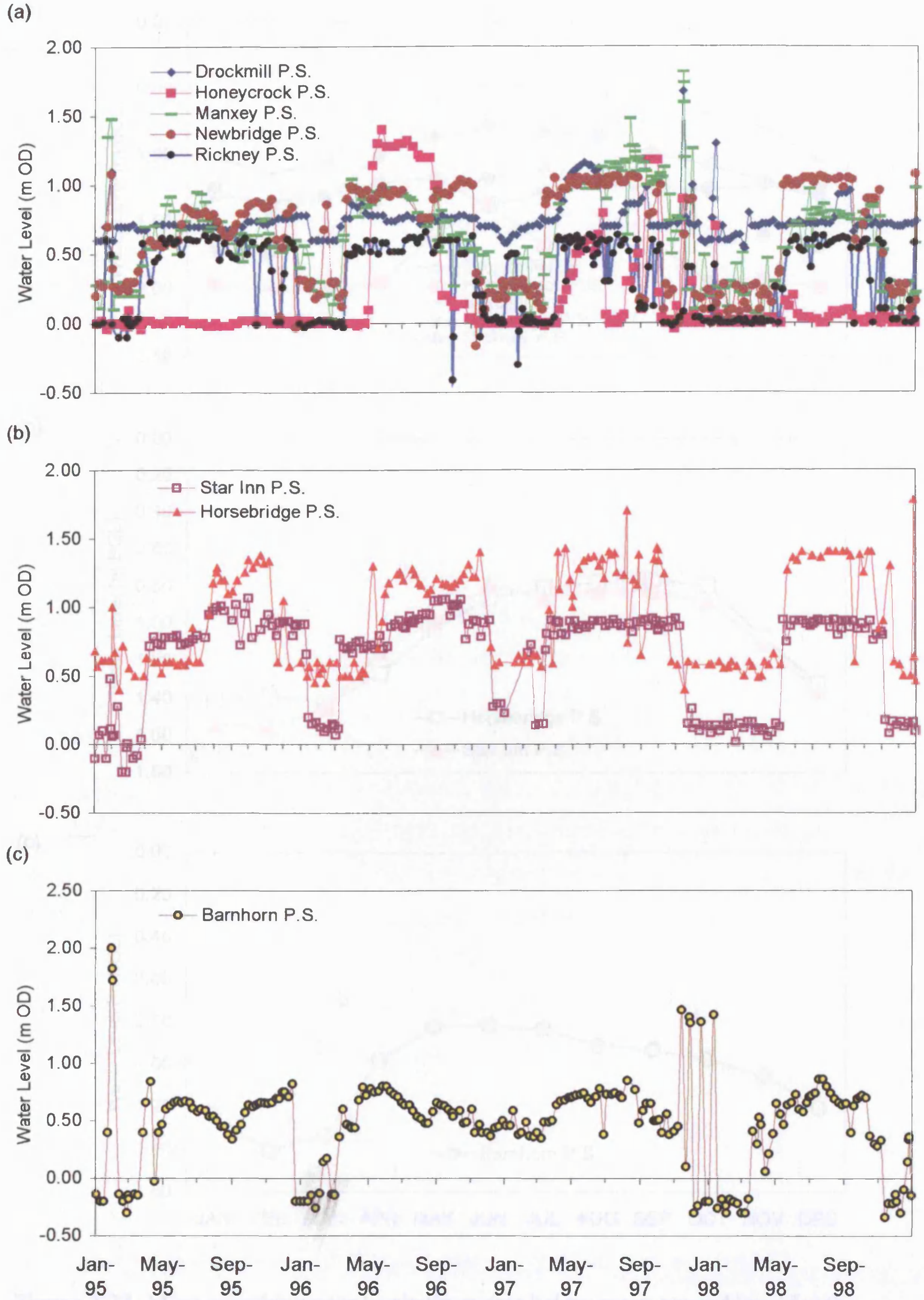


Figure 3.22. Weekly water levels in pumped channels (1995-1998) discharging to the (a) Pevensey Haven, (b) Wallers Haven and (c) East Stream embanked channel systems.

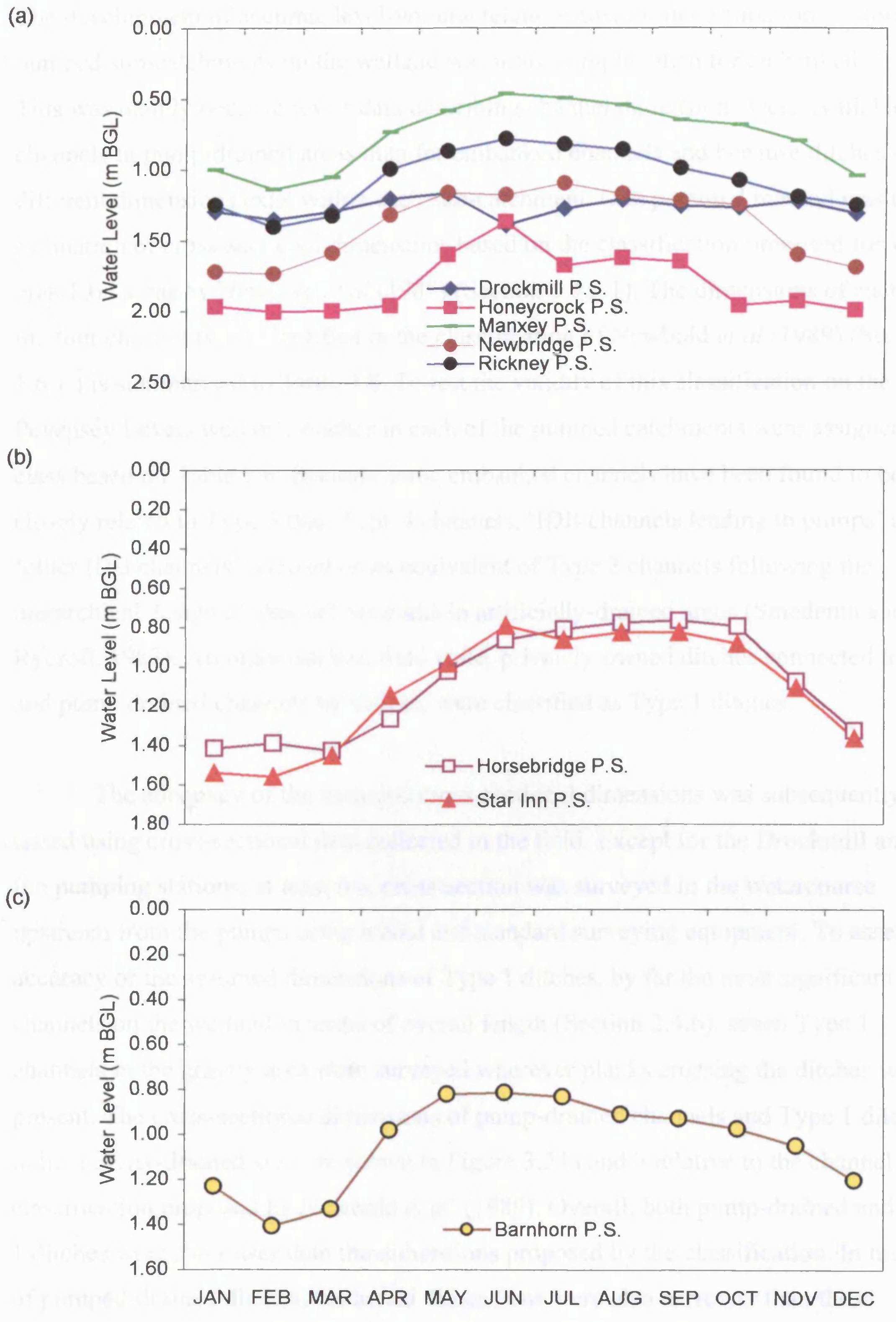


Figure 3.23. Mean monthly water levels (in metres below mean ground level for the period between 1995 and 1998) for pumping stations discharging to (a) the Pevensey Haven, (b) the Wallers Haven and (c) the East Stream embanked channel systems.

3.5.2. DITCH DIMENSIONS

The development of accurate level-volume relationships for the estimation of storage in pumped sub-catchments on the wetland was more complex than for embanked channels. This was mainly because fewer data describing channel dimensions were available for channels in pump-drained areas than for embanked channels and because ditches of different dimensions exist within each sub-catchment. One potential method was the estimation of cross-sectional dimensions based on the classification proposed for wet grassland areas by Newbold *et al.*(1989) (Section 3.4.3.1). The dimensions of each of the four channel types identified in the classification of Newbold *et al.*(1989) (Section 1.6.1.) is summarised in Table 3.8. To test the validity of this classification on the Pevensey Levels wetland, ditches in each of the pumped catchments were assigned to a class based on Table 2.6. Because some embanked channels have been found to be more closely related to Type 3 than Type 4 channels, 'IDB channels leading to pumps' and 'other IDB channels' were taken as equivalent of Type 2 channels following the hierarchical design of channel networks in artificially-drained areas (Smedema and Rycroft, 1983). All other ditches, field scale, privately-owned ditches connected to IDB and pump drained channels by sluices, were classified as Type 1 ditches.

The adequacy of the assumed cross-sectional dimensions was subsequently tested using cross-sectional data collected in the field. Except for the Drockmill and Star Inn pumping stations, at least one cross-section was surveyed in the watercourse upstream from the pumps using a boat and standard surveying equipment. To assess the accuracy of the assumed dimensions of Type 1 ditches, by far the most significant channels on the wetland in terms of overall length (Section 2.4.6), seven Type 1 channels in the gravity area were surveyed wherever planks crossing the ditches were present. The cross-sectional dimensions of pump-drained channels and Type 1 ditches in the gravity-drained area are shown in Figure 3.24a and b relative to the channel classification proposed by Newbold *et al.* (1989). Overall, both pump-drained and Type 1 ditches were shallower than the dimensions proposed by the classification. In the case of pumped-drained ditches, the actual dimensions were also narrower than those proposed by Newbold *et al.* (1989). In contrast, Type 1 ditches were wider.

For Type 1 ditches, the wider and shallower cross-sections could be attributed to bank erosion due to trampling by stock, and sedimentation in the deepest areas of the ditch leading to a more triangular form than the trapezoidal shape suggested by

Newbold *et al.*(1989) (Figure 3.24.b). Two distinct classes of field scale ditch (Type 1) were evident from cross-sectional data. Dimensions depended on planar profile: ditches appearing rectilinear on 1:25,000 O.S. maps (CS7 and CS8 in Figure 3.24.b) were three metres wide, narrower and shallower than the remaining ditches. Sinuous ditches tended to be wider (four metres) and deeper than linear ditches. These differences are probably attributable to the age of each class: Middle Age drainage ditches followed the natural drainage lines of the primary marsh, contrasting with rectilinear drains of the 17th to 19th Centuries (Cook, 1994) (Section 1.6.1). Sinuous ditches were more closely related to the Type 1 ditch dimensions proposed by Newbold *et al.* (1989) than sinuous channels. Linear and sinuous ditches had mean cross-sections 50% and 73% respectively of the Type 1 dimensions proposed by the classification.

Cross sectional data for pump-drained channels replicated trends for Type 1 ditches. Two contrasting types of pump-drained channel were evident. All ditches were shallower than Type 2 ditches and in the case of drains leading to the Barnhorn, Rickney and Newbridge pumping stations, channels were also narrower (Figure 3.24.a). These ditches were only 5 metres wide compared to the 8-metre width proposed by the classification. For remaining pumped drains, the width proposed by Newbold *et al.* (1989) for Type 2 ditches was coincident with actual channel width. In all cases however, total cross sectional areas of the ditches were smaller than those proposed by the classification. The cross-sectional areas of Type 1 – 4 ditches are shown in Table 3.8. For the Barnhorn, Rickney and Newbridge cross-sections, areas were more closely related to Type 1 than Type 3 ditches, and were 115%, 115% and 153% of Type 1 ditch dimensions. For remaining pump-drained ditches, areas were between 45 and 71 % of Type 2 ditches.

The differences between actual cross-sectional dimensions and those proposed by Newbold *et al.* (1989) provided an indication of potential inaccuracies introduced to surface water storage calculations if these data had been used in water balance calculations without verification. The results indicate that field measurement of cross-sectional dimensions should be undertaken at wetland sites where the ditch classification is applied. This will ensure that different ditch classes within a wetland can be assigned to the appropriate Newbold *et al.* (1989) ditch type, as this relationship is likely to vary from site to site.

	Description	Width (m)	Depth (m)	CSA (m^2)
Type 1	Small Private ditches	3.0	1.5	3.0
Type 2	IDB watercourses	8.0	3.0	14.0
Type 3	Intermediate IDB channels leading to pumping stations	10.0	3.0	32.0
Type 4	Embanked carrier drains	20.0	5.0	69.0

Table 3.8. Description and dimensions of the ditch classification proposed for wet grasslands in the UK (Newbold *et al*. 1989).

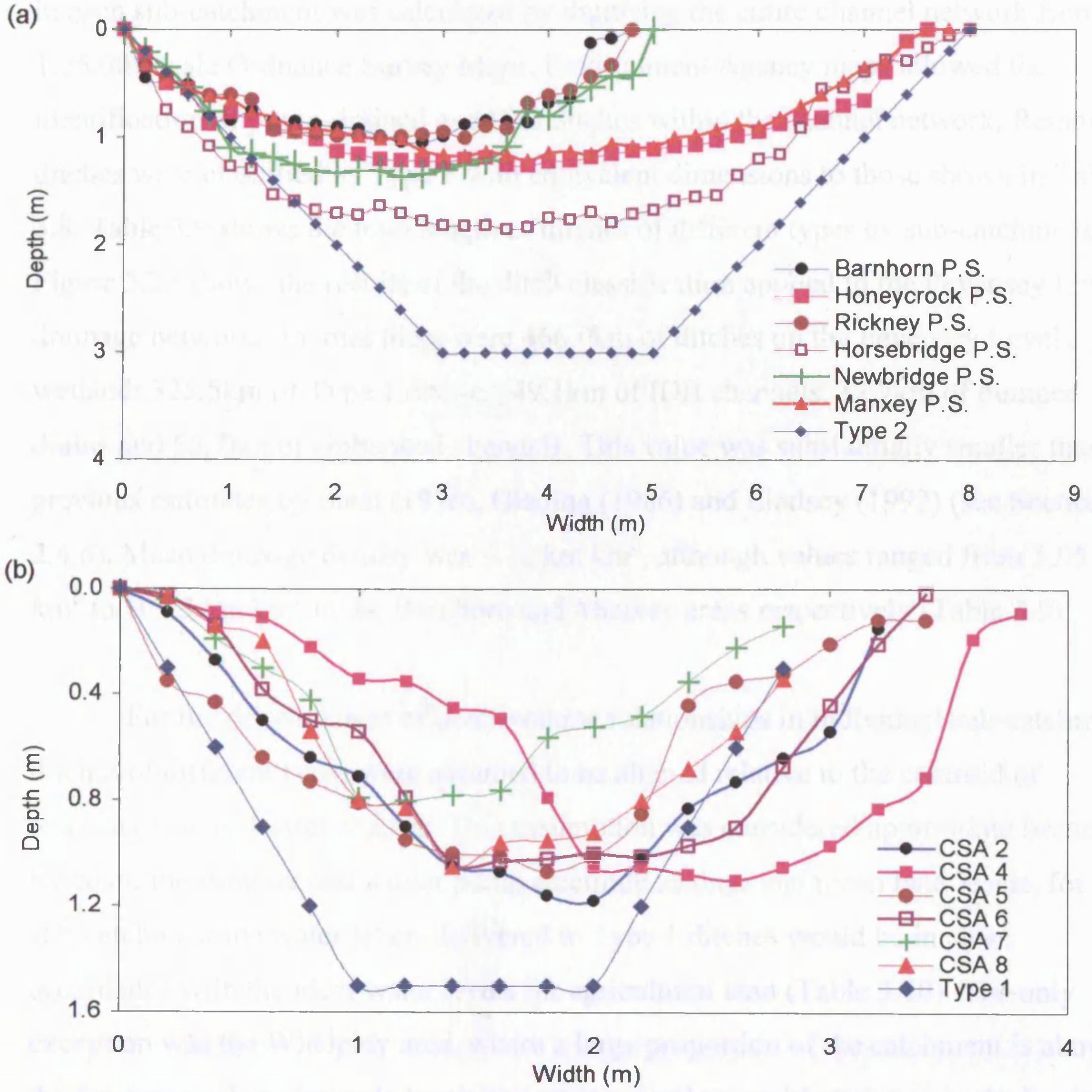


Figure 3.24. Actual cross-sectional dimensions on the Pevensey Levels relative to those proposed by Newbold et al. (1989) for (a) pumped drains and (b) Type 1 ditches.

3.5.3. SURFACE WATER STORAGE

Given the inaccuracies of the Newbold *et al.* (1989) classification in the local context, level-volume relationships for each pumped sub-catchment were established by combining available cross-sectional data with data describing the total channel length within each sub-catchment. In the absence of more detailed data, it was necessary to assume that the cross-sectional dimensions of pumped drains were uniform throughout their length. In the case of pumped drains associated with the Manxey, Newbridge, Barnhorn, Honeycrock, Rickney and Horsebridge pumping stations, the cross-sectional data employed were those shown in Figure 3.24.a. For pumped ditches connected to the Star Inn and Drockmill pumping stations, dimensions were taken as equivalent to Type 2 ditches in the classification provided by Newbold *et al.* (1989). The total ditch length in each sub-catchment was calculated by digitising the entire channel network from 1:25,000 scale Ordnance Survey Maps. Environment Agency maps allowed the identification of pump-drained and IDB ditches within the channel network. Remaining ditches were classified as Type 1 with equivalent dimensions to those shown in Table 3.8. Table 3.9 shows the total length of ditches of different types by sub-catchment. Figure 3.25 shows the results of the ditch classification applied to the Pevensey Levels drainage network. In total there were 466.0km of ditches on the Pevensey Levels wetland: 325.5km of Type 1 ditches, 49.1km of IDB channels, 32.7km of pumped drains and 58.7km of embanked channels. This value was substantially smaller than previous estimates by Steel (1976), Glading (1986) and Lindsey (1992) (see Section 2.4.6). Mean drainage density was 9.72 km km^2, although values ranged from 5.05 km km^2 to 10.23 km km^2 in the Barnhorn and Manxey areas respectively (Table 3.9).

For the development of level-volume relationships in individual sub-catchments, ditches of different types were assumed to be aligned relative to the centroid of respective cross-sectional areas. This assumption was considered appropriate because, based on the summer and winter pump electrode settings and mean land levels, for each sub-catchment the water levels delivered to Type 1 ditches would be in close accordance with the ideal water levels for agricultural land (Table 3.10). The only exception was the Whelpley area, where a large proportion of the catchment is above the 5m contour but electrode levels are set in accordance with drainage in the lowland parts of the catchment. As a result a level of 2.0m OD was used for this catchment, which is approximately the mean land level of the Whelpley lowland area (Mick Philips, Sluicekeeper, Pers. Comm.).

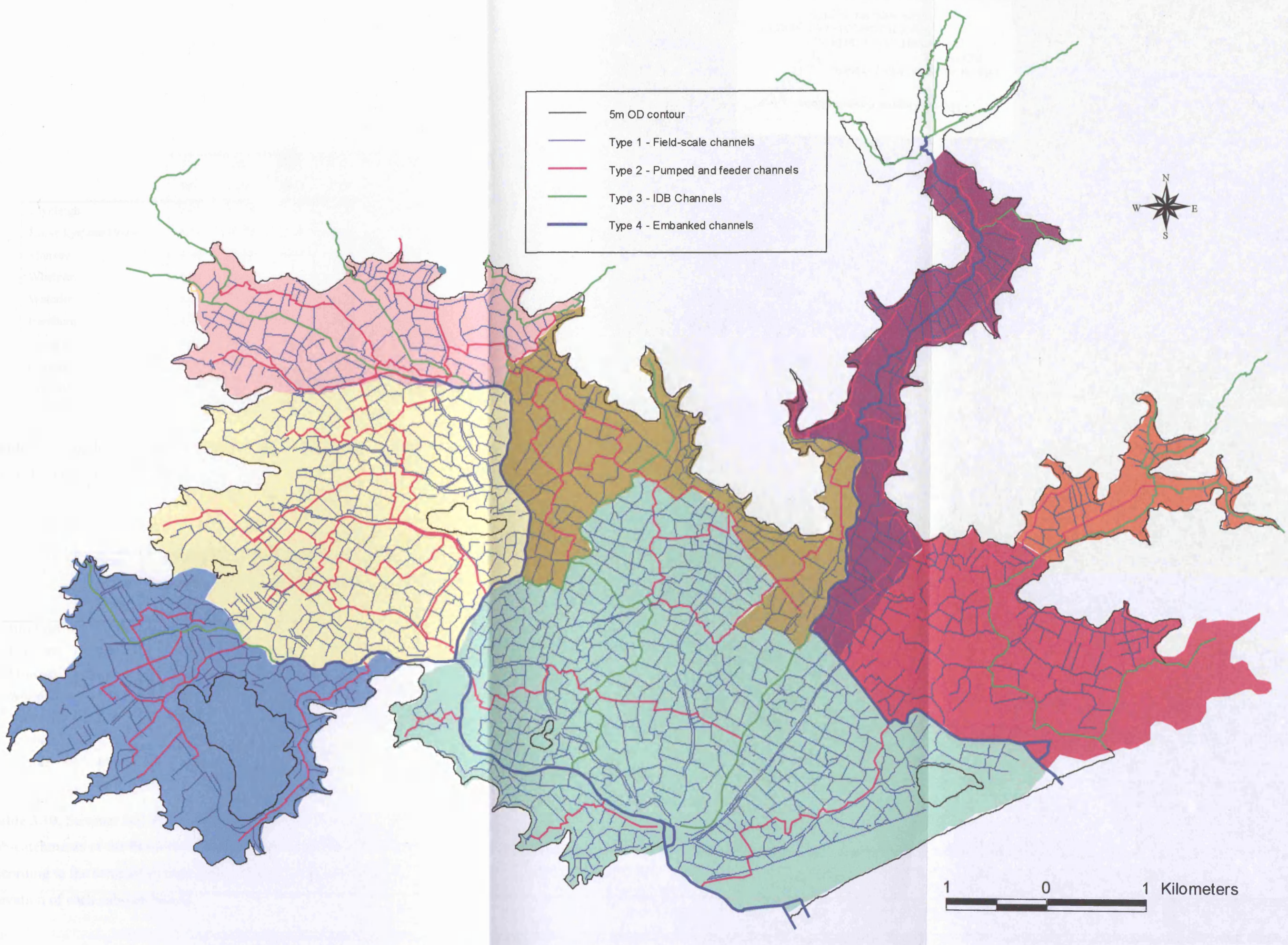


Figure 3.25. The ditch network of the Pevensey Levels by sub-catchment, showing embanked, IDB and pumped channels, and Type 1 ditches.

	Area (km^2)	Type 1 (km)	IDB (km)	Pumped (km)	TOTAL (km)	Density (km/km^2)
Glynleigh	6.07	13.74	7.25	2.62	23.61	7.77
Horse Eye and Down	8.34	65.14	5.23	6.04	76.41	9.16
Manxey	4.16	28.81	8.98	4.76	42.55	10.23
Whelpley	4.03	30.07	3.30	2.64	36.01	8.94
Waterlot	3.34	15.49	3.70	8.17	27.36	8.19
Barnhorn	1.40	6.22	0.00	0.86	7.08	5.05
Star Inn	5.56	33.32	9.01	4.47	46.80	8.42
Gravity	14.79	118.96	9.57	0.00	128.53	8.69
TOTAL	47.70	325.49	49.12	32.68	405.21	9.72

Table 3.9. Lengths of ditches of different types in each of the pumped sub-catchments on the Pevensey Levels wetland.

	Mean Elevation (m OD)	SUMMER Electrode Level (m OD)	WINTER Electrode Level (m OD)	SUMMER Water level (m BMFL)	WINTER Water level (m BMFL)
Glynleigh	2.00	0.60	0.40	0.65	0.85
Horse Eye	2.00	0.60	0.20	0.65	1.05
Manxey	1.40	0.00	-0.15	0.65	0.80
Whelpley	3.50	0.60	0.13	2.15	2.62
Waterlot	2.00	0.30	0.30	0.95	0.95
Barnhorn	1.50	0.25	-0.03	0.50	0.78
Star Inn	1.75	0.60	0.25	0.40	0.75

Table 3.10. Summer and winter water levels delivered to Type 1 ditches in pumped sub-catchments of the Pevensey Levels. Based on the alignment of different ditch types according to the centroid of their cross-sectional area and the mean field surface elevation of each sub-catchment.

Level-volume relationships for pump-drained sub-catchments on the Pevensey Levels are shown graphically in Figure 3.26. The regression coefficients required to compute volumetric storage from water level data at individual pumping stations are detailed in Table 3.11. For each sub-catchment, volumetric storage during the water balance period was calculated based on the assumption that at the time of measurement, water levels at the pumping station were effective over the entire pumped sub-catchment. This assumption relies on the rational behaviour by farmers, dictating that landowners within each sub-catchment will always be connected to the pumped drain because pumping stations provide the water levels suitable for maximum agricultural productivity on the wetland. Although this assumption undoubtedly simplifies the actual management of the ditch system, a more spatially distributed approach would have greatly increased the data requirements of the water balance calculation, and in any case these data were not available. Chapter 5 describes a method for the estimation of volumetric storage in field scale (Type 1) ditches. Using the level-volume relationships for the pump-drained sub-catchments, total surface water storage in the pumped areas of the wetland could be calculated by

$$S_{\text{Pumped Catchments}} = S_{\text{Horse Eye}} + S_{\text{Waterlot}} + S_{\text{Manxey}} + S_{\text{Star Inn}} + S_{\text{Whelpley}} + S_{\text{Glynleigh}} + S_{\text{Barnhorn}} \quad (3.21)$$

where S denotes surface water storage (m^3) and the subscript identifies the pumped sub-catchment. Results suggested that bankfull storage in the pump-drained lowland ditch network was considerable, and equivalent to 3.10 million m^3 (Table 3.12). Assuming that all incident rainfall would contribute to ditch storage, it would take 145mm of rain to fill ditches in pumped areas of the wetland. This was approximately equivalent to the total mean monthly rainfall of January and February at Horseye during the period 1961-1990 assuming that 100% of catchment rainfall is conveyed to ditches. Results illustrated the importance of lowland surface water storage on the wetland, especially since the data shown do not include storage in embanked channels or the gravity-drained area, which is the largest sub-catchment on the wetland. Surface water storage in the gravity area is considered in Section 3.6.5. Differences between the actual volumetric storage in the ditch network and the volumes required to achieve bankfull conditions are shown in Figure 3.27. The extent of these differences highlight the potential difficulties posed by the implementation of revised water level management strategies across the wetland, an issue that is further considered in Section 3.7.2.

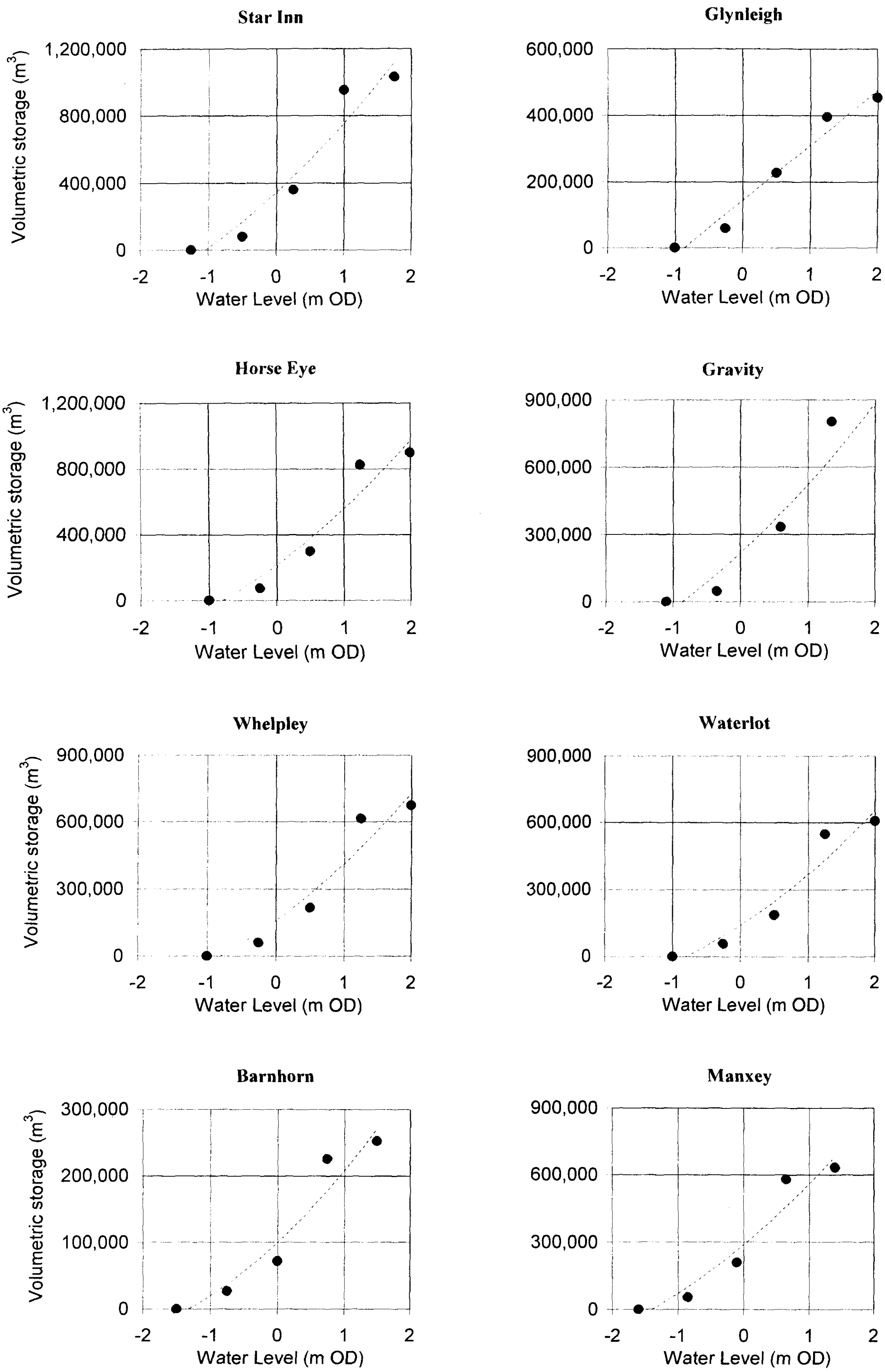


Figure 3.26. Graphical representation of the level-volume relationships for pumped sub-catchments of the Pevensey Levels wetland.

	Storage (m^3) = a(water level)^2 + b(water level) +c		
	a	b	c
Glynleigh (Drockmill & Honeycrock P.S.)	-2 x 10^{-10}	165,940	143,830
Horse Eye (Rickney P.S.)	38,171	301,309	216,200
Manxey (Manxey P.S.)	27,219	244,234	288,445
Whelpley (Newbridge P.S.)	30,590	223,110	159,493
Waterlot (Horsebridge P.S.)	29,657	197,653	140,640
Barnhorn (Barnhorn P.S.)	13,680	93,762	100,017
Star Inn (Star Inn P.S.)	40,194	371,385	343,500

Table 3.11. Level-volume relationships for the calculation of storage in sub-catchments on the Pevensey Levels wetland based on the water level at the pumping station.

Sub-Catchment	Type 1 (million m^3)	IDB Channels (million m^3)	Pumped Channels (million m^3)	TOTAL (million m^3)
Glynleigh	0.02	0.04	0.19	0.252
Horse Eye	0.30	0.13	0.17	0.60
Manxey	0.21	0.13	0.08	0.42
Whelpley	0.19	0.08	0.16	0.43
Waterlot	0.14	0.03	0.20	0.37
Barnhorn	0.04	0.08	0.24	0.36
Star Inn	0.34	0.16	0.16	0.66
TOTAL	1.24	0.67	1.19	3.10

Table 3.12. Volumetric storage at bankfull conditions for pumped sub-catchments on the Pevensey Levels wetland. Bankfull conditions refer to water levels equivalent to the mean elevation of field in each sub-catchment shown in Table 3.10.

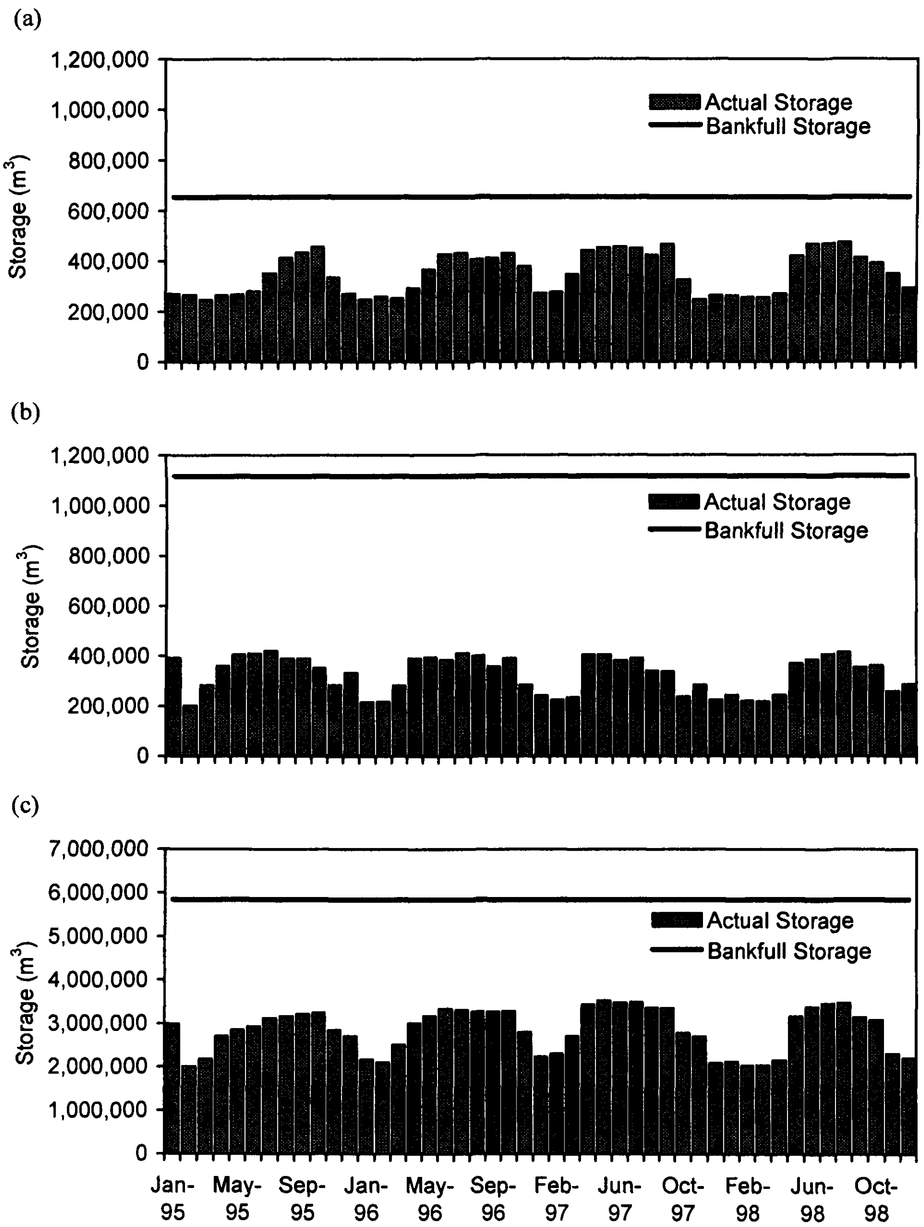


Figure 3.27. Mean monthly volumetric storage between 1995 and 1998 in the (a) Star Inn sub-catchment, (b) Waterlot sub-catchment and (c) all pumped sub-catchments relative to bankfull storage.

3.5.4. PUMP FUNCTIONING

Analysis of pump hour data for the Pevensey Levels illustrated the difficulties posed by predicting the volumes of water pumped from the wetland based on simple hydrological parameters, such as total rainfall. Although for most pumping stations a distinct trend was apparent when annual pumped volumes and rainfall were compared (Figure 3.28), on a monthly basis, a direct relationship was not apparent for any of the pumping stations considered in the analysis (Figure 3.29). Results could be ascribed to both the seasonality of pumping station operation, and the fact that under natural behaviour, the proportion of the catchment contributing to runoff, and hence the volume pumped, will vary not only depending on incident rainfall, but also on precedent rainfall, soil moisture characteristics and the characteristics of the rainfall event (Beran, 1987). Management practices within the sub-catchment also need to be considered.

The proportion of monthly rainfall pumped from individual pump-drained sub-catchments on the wetland between 1984 and 1998 is shown in Figure 3.30. For all pumping stations, the proportion of rainfall pumped was greatest in the winter and smallest in the summer. This corresponds with traditional models of catchment hydrological behaviour, where incident rainfall replenishes soil and surface water storage during dry periods, and larger magnitudes of runoff are generated during wet periods when surface and groundwater stores are close to saturation (Shaw, 1993). For most pumping stations, the proportion of summer rainfall pumped was within the range of Standard Percentage Runoff (SPR) figures proposed by Binnie and Partners (1988) for the Willingdon Level (10 - 47%), located close by, and those proposed by Beran (1987) for Newborough Fen (Table 1.10). In the case of some pumping stations however, the volume pumped exceeded incident rainfall. Two distinct types of catchment could be identified on the Pevensey Levels relative to the proportions of rainfall pumped. Whilst the volumes pumped from the Barnhorn, Star Inn and Glynleigh catchments (Figure 3.30 a-d) rarely, if ever, exceeded 100% of monthly rainfall, this was a common feature for the remaining catchments, especially during the winter months (Figure 3.30 e-h). In the case of the Horseye and Down catchment, this feature could be ascribed to the influence of the Hailsham South STW, which discharges to it. Data suggested that a large proportion of the water pumped by this station originated from the works. In the summer the proportions of rainfall pumped were considerably greater than those recorded at other pumping stations with potentially important implications for the water balance and water quality in the western part of the wetland.

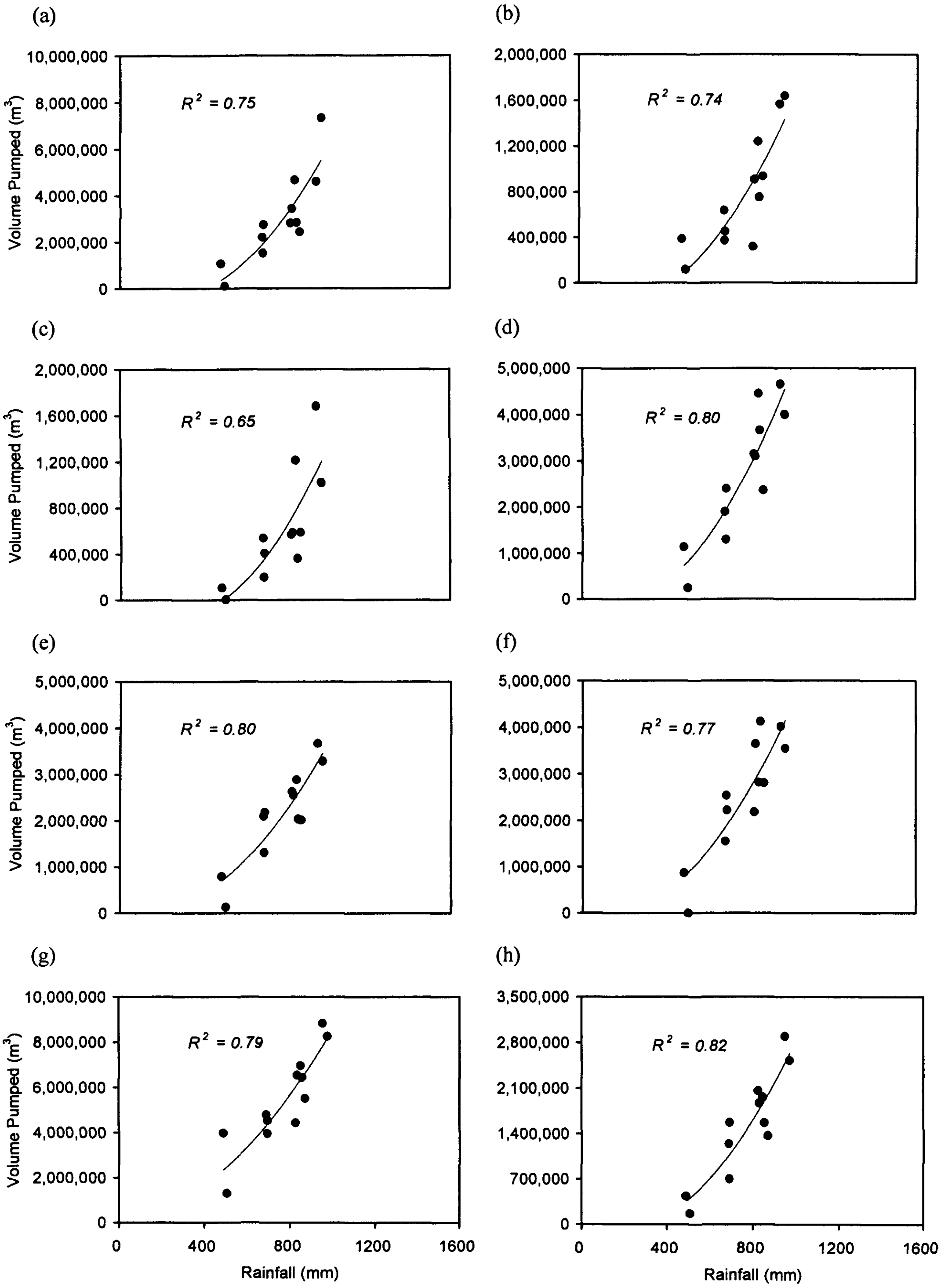


Figure 3.28. The relationship between annual rainfall and the volume pumped from the catchments of the (a) Honeycrock, (b) Drockmill, (c) Barnhorn, (d) Horsebridge, (e) Manxey, (f) Newbridge, (g) Rickney and (h) Star Inn pumping stations.

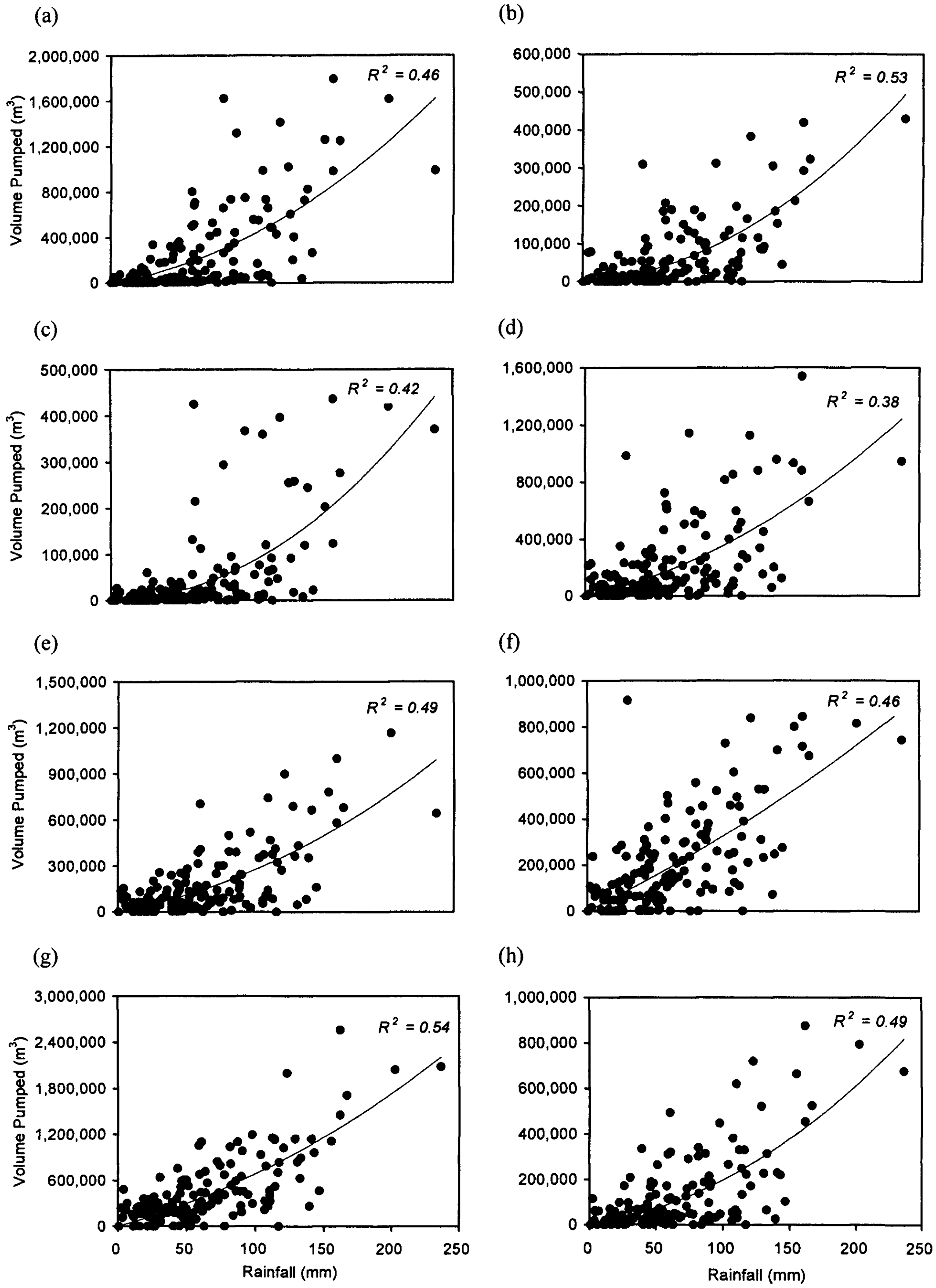


Figure 3.29. The relationship between monthly rainfall and the volume pumped from the catchments of the (a) Honeycrock, (b) Drockmill, (c) Barnhorn, (d) Horsebridge, (e) Manxey, (f) Newbridge, (g) Rickney and (h) Star Inn pumping stations.

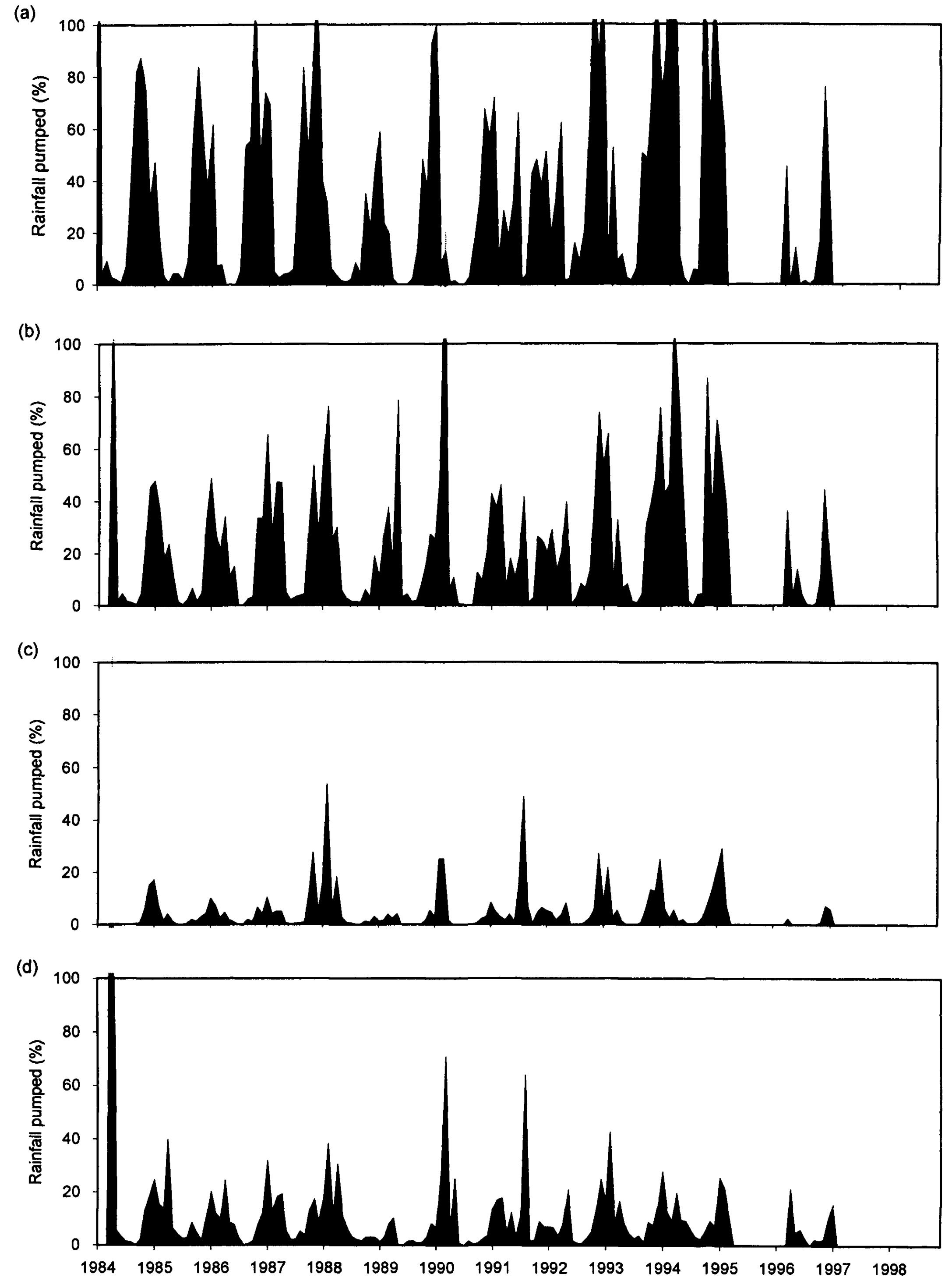


Figure 3.30. The proportion of monthly rainfall pumped from the sub-catchments of the Pevensey Levels wetland between 1984 and 1998 by pumping stations at (a) Honeycrock, (b) Drockmill, (c) Barnhorn and (d) Star Inn.

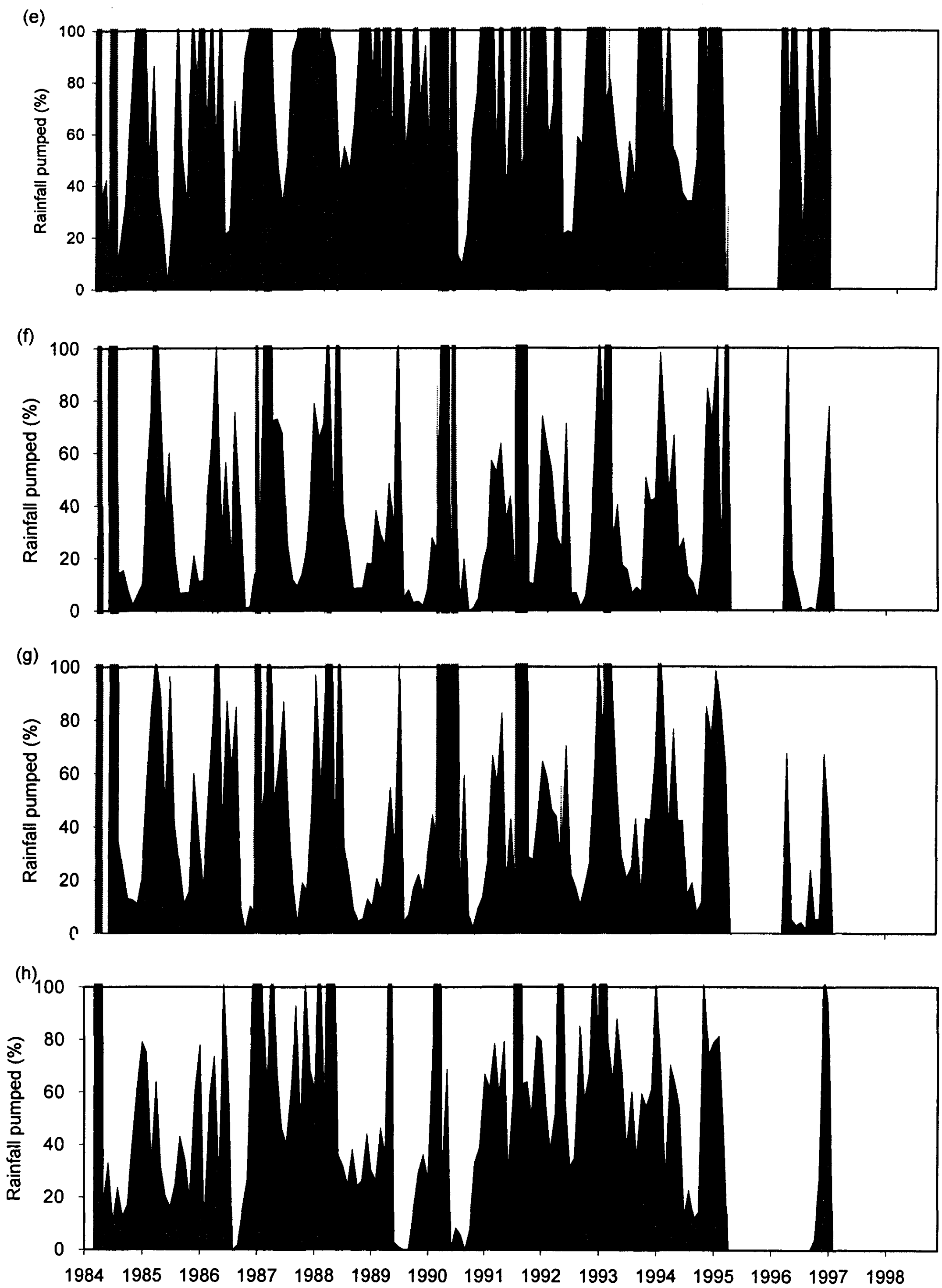


Figure 3.30. The proportion of monthly rainfall pumped from sub-catchments of the Pevensey Levels wetland (1984-1998) from (e) Rickney, (f) Horsebridge, (g) Manxey and (h) Newbridge.

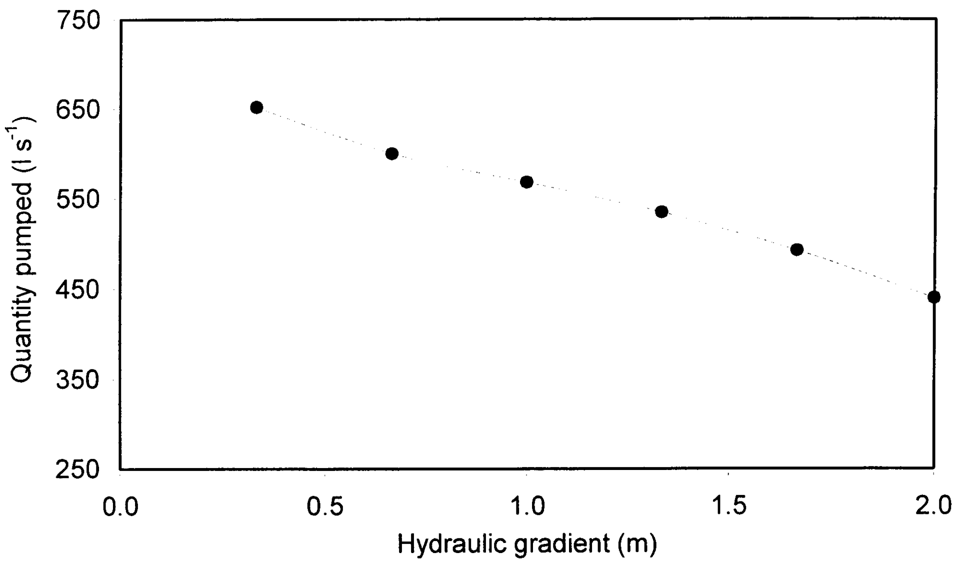


Figure 3.31. Efficiency of the Manxey pumping station relative to the hydraulic head (Peter Blackmore, Flood Defence Section, EA, Pers. Comm.).

For the remaining catchments, the reasons for the differing relationships apparent between rainfall and proportion of rainfall were less clear. Hypotheses included the temporal resolution of the data, inaccuracies in catchment delineation and pump efficiency considerations. Runoff generated by rainfall falling at the end of one month could have been recorded as pumping in the next month, causing the proportion of rainfall pumped to be exceptionally high, especially if the following month was dry. However, this hypothesis could be excluded because if true, such features would be coincident for all pumping stations, and they were not. Differences were most probably associated with pump efficiency considerations, particularly because there is not necessarily a well defined relationship between hour run and the volume pumped (Gilman, 1990), an assumption incorporated within the method employed to calculate the volumes pumped from the wetland (Equation 3.12). During pumping, the hydraulic head is continually changing due to variations in water level on both sides of the pumping station. For the Manxey pumping station, the effects on pump capacity of water level variations in the inflow and outflow channels (termed the hydraulic gradient), is shown Figure 3.31. Depending on the head between the pump inflow and outflow channel, pump capacity may decrease by up to 40 %, a feature identified by

Marshall (1989) and Ritzema (1994) for all pumping stations except those of archimedean type. Indeed, three of the four axial type pumping stations on the wetland (Horsebridge, Manxey and Newbridge) were those characterised by pumping that regularly exceeded rainfall (Figure 3.30.f-h respectively). The less frequent exceedances beyond rainfall recorded at Honeycrock, the only other axial type pump on the wetland, could also be ascribed to pump efficiency considerations, providing further support for the validity of this hypothesis.

Other factors are also likely to contribute to pumping station efficiency. The volume of water a pump can access, termed the backwater length (Beran, 1987), will vary throughout the year. Channel roughness is particularly prone to seasonal variations due to the growth of macrophytes within channels (Beran, 1987). For sluices on Llangofan Fen for example, Gilman (1990) has identified changes in the stage-discharge relationship due to seasonal growth and die-back of ditch vegetation. Seasonal variations in channel conveyance of 73% and 76% due to the growth of emergent ditch-bank vegetation and submerged vegetation respectively have also been reported for canals in Egypt by Bakry *et al.* (1992). Such processes are likely to be of importance on the Pevensey Levels wetland. Annual dredging to remove macrophytes is a crucial component of the maintenance of the drainage system, and has become especially important following the invasion of *Hydrocotyle rananculoides* in recent years (Brian Deeprose, Environment Agency, Pers. Comm).

The fore-mentioned factors highlight some of the limitations associated with the method advocated by Marshall (1989) for the quantification of pumping. Although this method provided a simple means for the calculation of volumes pumped from sub-catchments on the Pevensey Levels wetland, findings clearly suggest that water level data collected at a higher spatial and temporal resolution in both pump-drained and pump-outflow watercourses will be required, especially if axial-type pumps are present. Results highlight the need for the continued collection of data describing pump functioning by the operating authority on the Pevensey Levels, especially given the influence of pumping on estimates of losses to sea. Based on the results presented in this section, some inaccuracies in the wetland water balance were therefore expected.

3.6. Hydrology at the field scale

To date, the historical assessment of hydrological functioning of the Pevensey Levels at the field scale has been less intensively considered than that at the pump-unit or catchment scale. This is mainly because at the field scale, owners have the primary responsibility for flood defence on their land. This has resulted in a general lack of monitoring at this scale by the IDB, since at both the catchment and pumped-unit scales, the primary reason for data collection has been the regulation of IDB operations. Increasingly however, the introduction of schemes for the ecological enhancement of these habitats, has resulted in a more pressing need to understand the hydrological dynamics of wet grasslands at a more reduced spatial scale. This is clearly illustrated by the case of the Pevensey Levels, where the installation of field-scale monitoring has been a direct response to the need to establish the effectiveness of raising ditch water levels in providing the in-field conditions suitable for typical wet grassland biota (Section 2.8.4). Indeed, the field scale monitoring network was installed on what has been termed the WES 'Pilot Area' one of the first areas where raised water levels were a component of management.

Preliminary details regarding the components of the field-scale monitoring network on the Pevensey Levels have been previously considered in Section 2.8.4. Figure 3.32 provides a more detailed account of the location of the various components of the field-scale hydro-meteorological monitoring network installed on the Sussex Wildlife Trust (SWT) National Nature Reserve (NNR). Three fields were initially instrumented, two within the NNR where water levels are maintained for nature conservation (Field Two and Field Three) and one in an area employed predominantly for grazing (Field One). In each field, dipwells were installed in the clay at increasing distances from the ditch (Section 2.8.4). 12 dipwells were installed in Field One, 13 in Field 2 and 11 in Field 3. Three piezometers monitoring water levels in the peat layer were also installed in each field. These are located adjacent to dipwells in the centre of each field and at a distance of 8m from the ditch at either end of each transect. Field Three however has only two piezometers as at one end of that transect, the peat layer could not be found (Douglas and Hart, 1994). Between January 1995 and December 1998, water table elevation in all dipwells and piezometers was measured on a roughly fortnightly basis, and water level recorder charts providing daily estimates of ditch water levels were changed at the same time. However, the Field One monitoring network was decommissioned in November 1996, when water level management in that area came under the scrutiny of WES. The bulk of the analysis presented in this section therefore refers to Fields Two and Three, although data collected prior to November 1996 are employed to compare the hydrology of areas managed for nature conservation and agriculture on the wetland.

In combination, data from this monitoring network provided a powerful tool to investigate the interaction between ditches and the shallow groundwater on the Pevensey Levels wetland, a relationship that is crucial in terms of both agriculture and nature conservation in wet grasslands (Section 1.6). Indeed, the collection and analysis of hydrological data collected from this monitoring network formed a central component of the work presented in this thesis. In this section, these data are used to evaluate the hydrological functioning of the wetland at the field scale, an issue that is extended in Chapter Five. A particular focus is on the effects of raising ditch water levels on in-field water table levels. One of the main criticisms of current wetland restoration scheme raised by conservationists is that WES prescriptions are insufficient to promote the conditions for wet grassland biota of conservation importance on field surfaces (Table 2.14). Conclusions from such analyses are also of central bearing to the farming community because they provide an indication of the likely impacts of schemes such as the WES on agricultural practices on the wetland.

Complimentarily to the assessment of field scale hydrological functioning to address stakeholder concerns, field scale data also allow the quantification of some of the remaining variables required for the computation of the catchment-wide water balance. For example, water level data collected from the monitoring network are employed to quantify storage in channels of the gravity area, a means of complementing estimates of surface water storage in the pumped sub-catchments and embanked channels of the wetland. Water table data have been used to estimate ground- surface water interactions on the wetland, and to inform the way in which this process is incorporated within water balance calculations. Indeed, water table data have not been used directly in the wetland water balance calculation as water table levels on the SWT Reserve are not necessarily representative of the wetland as a whole. Consequently they have been used to give an indication of the magnitude of the process only. Soil and groundwater storage have been subsequently calculated using the soil moisture deficit data provided by the Met Office MORECS system (Figure 3.2) where required.

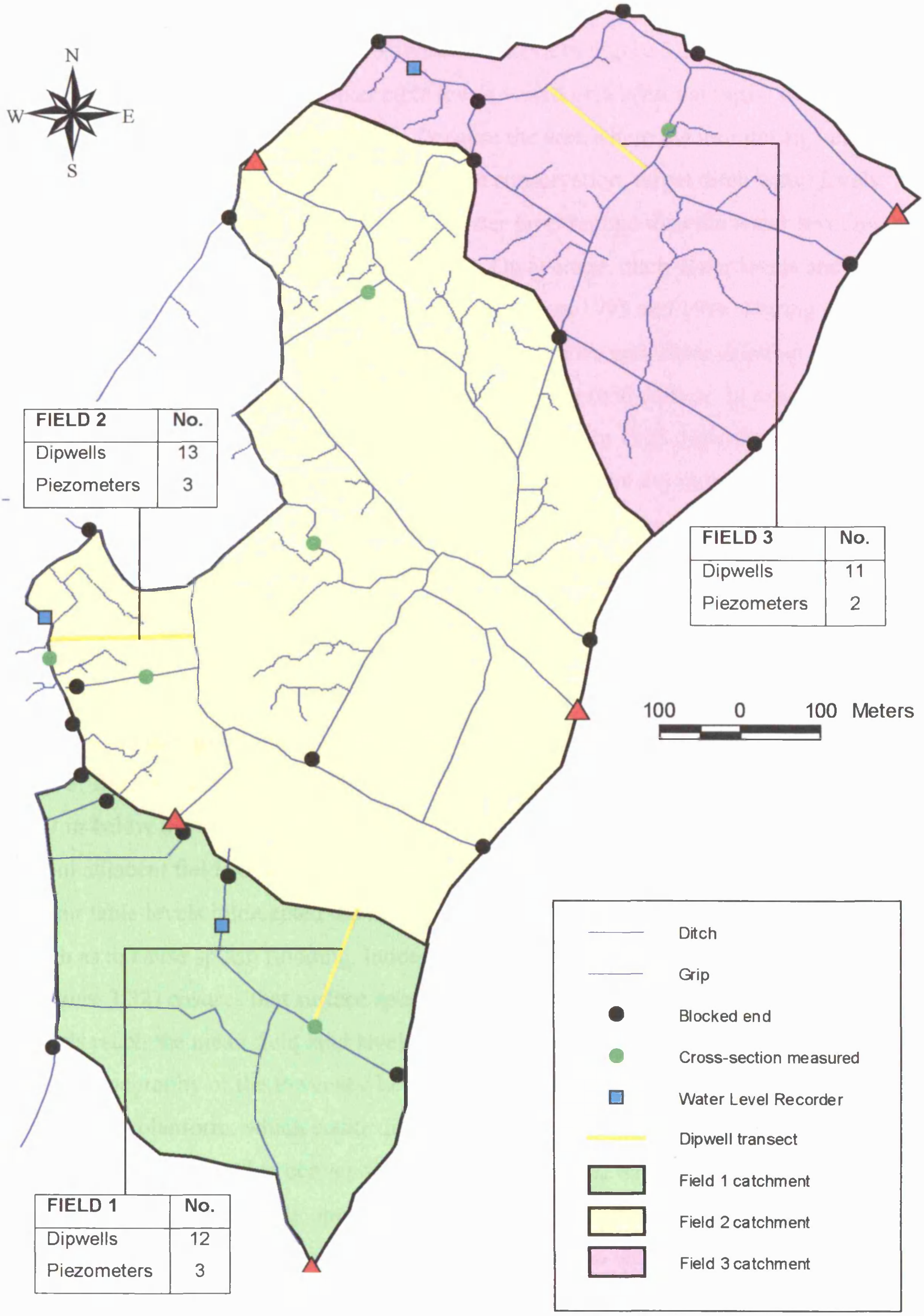


Figure 3.32. The hydrological monitoring network on the SWT Reserve.

3.6.1. TRENDS DURING THE STUDY PERIOD

For the water balance period, time series of ditch water level and field-centre water table levels for the three fields monitored are shown in Figure 3.33. Throughout the study period, both ditch and water table levels varied on a seasonal basis, with maxima in winter and minima in late summer. Because the area where the monitoring network was installed was a NNR managed for nature conservation, target ditch water levels were more in accordance with a 'natural' water level regime than the water level regime associated with agriculture (see Figure 1.11). On average, ditch water levels and water table levels were higher during 1997 and 1998 than in 1995 and 1996. During the summers of 1995 and 1996, most dipwells in Fields Two and Three dried up and the water table retreated to more than one metre below the field surface. In most cases, this was the maximum measurable depth to the water table. In 1995 dipwells remained dry between late June and early October. In 1996 dipwells were dry only during the month of August (Figure 3.33). Ditch water levels mirrored these trends; towards the end of the summers of 1995 and 1996 the ditches were effectively dry, although as with the dipwells, the period during which this state was maintained was longer in 1995 than 1996.

Conditions during 1995 and 1996 contrasted with those apparent in 1997 and 1998. During these years, in-field water table levels in Field Two were never more than 0.7 m below the field surface and ditch water levels never receded more than 0.55 m from adjacent field surfaces. On numerous occasions during 1997 and 1998 field centre water table levels intercepted the field surface and ditch water levels were sufficiently high as to cause splash flooding. Indeed, the extensive lengths of 'grips' on the Reserve (Figure 3.32) ensures that surface splashing of field surfaces occurred before ditch water levels reach the mean field land level. Grips are an integral component of the microtopography of the Pevensey Levels wetland, and are small channels, generally sinuous in planform, which create distinctive patterns on field surfaces. Although traditionally exploited to convey field runoff into ditches, where ditch water levels have been raised, grips currently operate in reverse fashion, allowing ditch water to access in-field areas. During field visits, snipe *Gallinago gallinago* were most frequently observed in these areas, suggesting that water table levels were '*less than 0.2 m from the field surface*', as suggested by Green and Robins (1993). *Juncus* spp., a species tolerant of inundation is also most frequently located close to grips on the SWT Reserve (Plate 4.3.a).

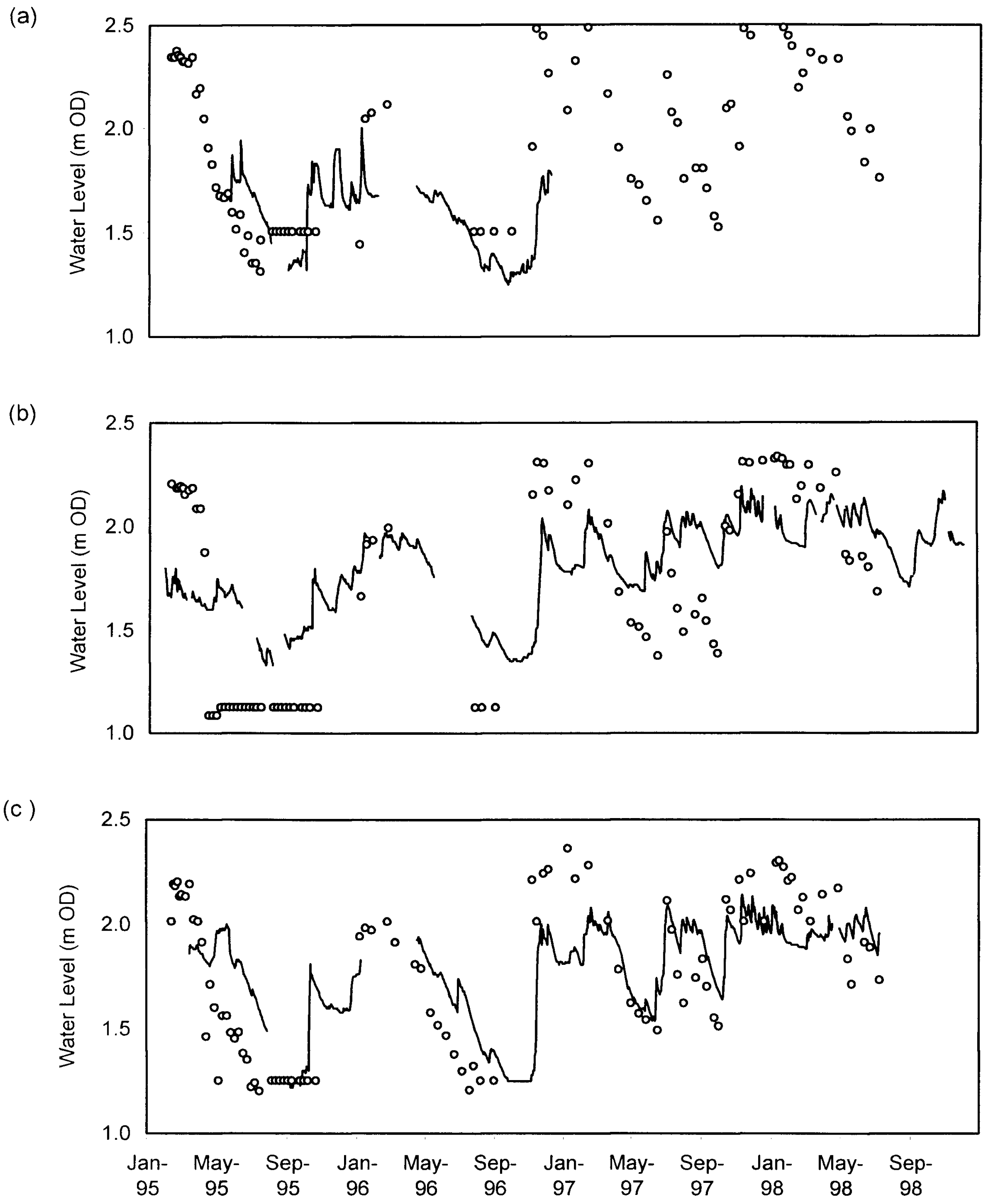


Figure 3.33. Ditch water levels (solid line) and field-centre water table levels (dots) on the SWT Reserve during the water balance study period: (a) Field One, (b) Field Two and (c) Field Three.

Data provided by the monitoring network provided an indication of the importance of the balance between rainfall and evapotranspiration on field-scale hydrology. Rainfall during the summers of 1997 and 1998 was considerably above average, whereas summer and annual rainfall totals for 1995 and 1996 were considerably less than long term averages (Section 3.3.1). These particularly low rainfall totals meant that during a large proportion of the summers of 1995 and 1996, water levels on the SWT Reserve were less than WES target water levels of '*no less than 0.3m below field level between January and August*' (Figure 3.34). During the summers of 1997 and 1998 (Figure 3.34), ditch water levels also fell below the minimum requirements associated with WES, although the duration of this period was considerably shorter than in 1995 and 1996. This was the case even though in July 1996, a new sluice was constructed on the reserve, raising the maximum attainable ditch water levels by 0.4m (Neil Fletcher, SWT Reserve Warden, Pers. Comm.). Results highlighted the difficulty of satisfying the water level requirements of the WES, even in the wetter years of 1997 and 1998, and were attributed to the enhanced evaporation and evapotranspiration rates that wetland surfaces promote. This issue is considered in detail in Chapter Four.

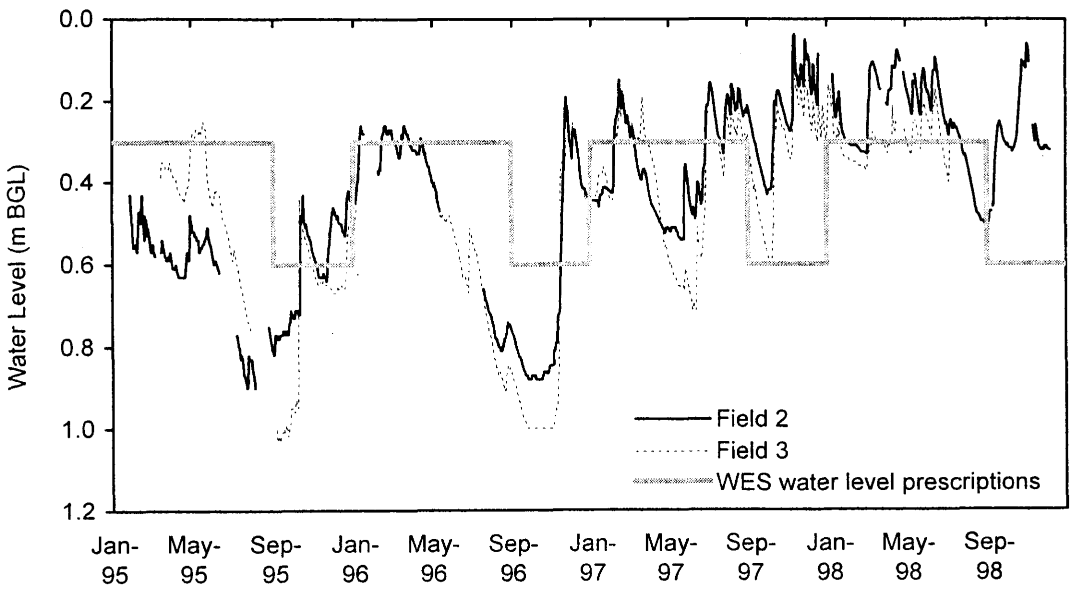


Figure 3.34. Ditch water levels in the gravity-drained area relative to WES water levels prescriptions during the water balance study period.

3.6.2. THE RELATIONSHIP BETWEEN DITCH AND FIELD WATER LEVELS

Calculations based on available estimates of hydraulic conductivity (*K*) for the site provided by Armstrong (1998) supported suggestions that water level management strategies such as WES would not deliver sufficiently different soil water regimes to those currently apparent wetland-wide. Based on the mean value of *K* obtained for the SWT Reserve (0.057 md^{-1}), it would take 1535 days for a water level set in a ditch to come into equilibrium with the water table in the centre of a field 175 metres wide, the approximate width of the fields monitored in the SWT Reserve. This was supported by detailed analyses of the relationship between ditch water levels and in-field water table levels. In most cases, the sphere of influence of the ditch was limited to dipwells located 2m and 5m from the ditch. This was supported by values of the coefficient of determination (R^2) obtained for the relationships between ditch water level and water table level in the dipwells closest to the ditches (Figure 3.35). At distances greater than 5m, water table variations seemed more closely related to the balance between rainfall and evaporation in the preceding period than to water levels in the ditch (Figure 3.35). In most cases, values of R^2 obtained from the relationship between water table level changes and the balance between rainfall and evapotranspiration in the preceding period were highest for dipwells at distances more than 5m from the nearest ditch (Table 3.13). The reverse was apparent for the relationship between water table level and ditch water level (Figure 3.35; Table 3.13).

In combination, these data suggest that the shallow groundwater on the wetland operated as a hydrological system largely independent of the ditches, a feature noted in other wet grassland sites in southern England dominated by clay substrates (Gavin, 2001). Results have important implications for the wetland management strategies associated with ditch water level prescriptions: attempts to achieve higher water table levels will probably require surface splashing as well as higher ditch water levels if providing more than wet ditch margins is to be a realistic management objective. On the Pevensey Levels, *K* potentially varies with depth (Figure 3.36) but, based on the mean value of K measured at the field surface (0.057md^{-1}), inundation water would take only 35 days to travel through 2.00m of clay, the typical thickness of clay on the wetland (Section 2.3). This result suggests that if shallow surface flooding can be maintained for over a month during winter, fully saturated soils in spring can be provided. This is a condition favoured by most species characteristic of wet grassland habitats (RSPB, ITE and EN, 1997).

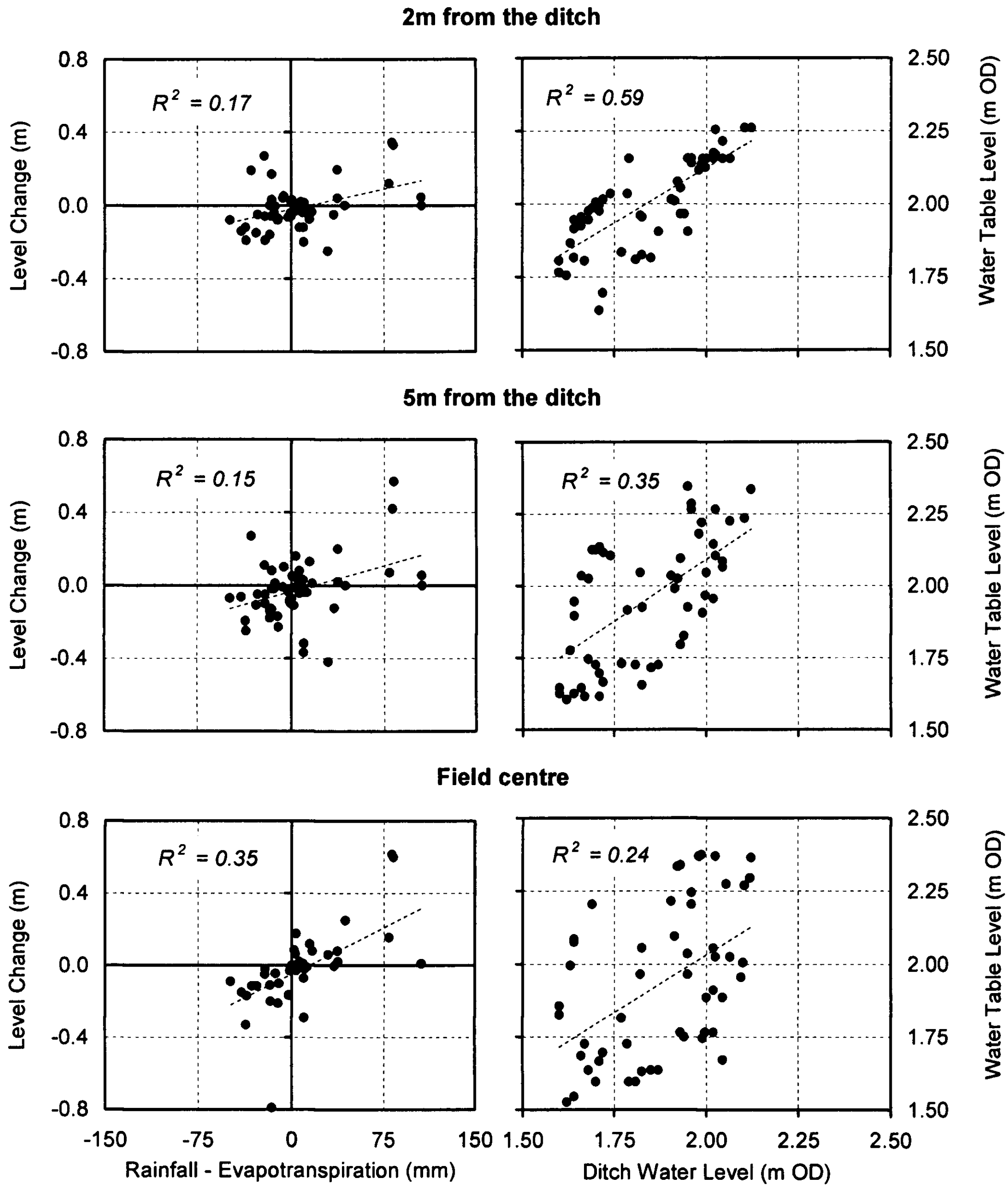


Figure 3.35. Relationship between ditch water levels and water table levels in Field Two for dipwells 2m from the ditch, 4m from the ditch and (c) in the field centre. Relationships between water table level change and the balance between rainfall and evapotranspiration are also shown.

DIPWELL NUMBER	FIELD 1			FIELD 2			FIELD 3		
	Distance from nearest ditch (m)	DWL	R - ET	Distance from nearest ditch (m)	DWL	R - ET	Distance from nearest ditch (m)	DWL	R - ET
1	2	0.22	0.03	2	0.60	0.28	2	0.61	0.34
2	5	0.25	0.38	5	0.43	0.43	5	0.50	0.22
3	10	0.27	0.54	10	0.18	0.48	10	0.61	0.41
4	20	0.28	0.44	20	0.11	0.40	20	0.53	0.47
5	40	0.24	0.62	40	0.15	0.35	40	0.45	0.46
6	85	0.20	0.52	60	0.16	0.35	75	0.42	0.40
7	55	0.23	0.50	100	0.19	0.19	40	0.52	0.40
8	40	0.34	0.61	60	0.20	0.60	20	0.60	0.43
9	20	0.36	0.52	40	0.13	0.50	10	0.61	0.42
10	10	0.34	0.19	20	0.17	0.43	5	0.62	0.46
11	5	0.20	0.15	10	0.31	0.36	3	0.45	0.31
12	2	0.14	0.24	5	0.15	0.15			
13				2	0.18	0.17			

Table 3.13. Coefficients of determination generated by (a) the relationships between ditch water levels and water table levels for all dipwells (DWL) and (b) the relationships between water table level change and the balance between rainfall and evapotranspiration in the preceding period (R-ET).

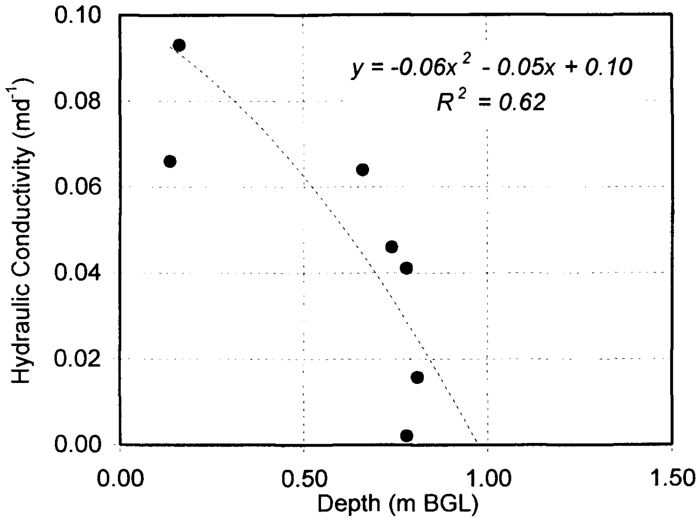


Figure 3.36. Variation of hydraulic conductivity with depth on the SWT Reserve. Water table level at the start of the pump test taken as an analogue of soil depth.

3.6.3. CATCHMENT_SCALE GROUND-SURFACE WATER INTERACTIONS

The limited evidence available for the interaction between ditches and fields on the Pevensey Levels suggested that for the calculation of the wetland water balance, seepage and phreatic contributions to the surface water components of local hydrology could be excluded. The adequacy of this assumption was tested by applying Darcy's Law to the Field Two ditch system where:

$$G_V = KIA \qquad \text{(Equation 3.21)}$$

where K is the hydraulic conductivity, taken as the mean of all the hydraulic conductivity samples (0.057 md^{-1}), I is the hydraulic gradient, calculated as the difference between ditch water level and water table levels in the field centre, and A is the area over which the exchange takes place. Based on previous work by Boelter (1967) and Miles (1980), A was calculated as a function of the hydraulic gradient since the largest area of exchange around an open ditch occurs where the hydraulic gradient is greatest. Based on a model of ditch water level variations on the SWT Reserve described in Section 5.3.3, A can be given by:

$$A = 2.5\, I\, L \qquad \text{(Equation 3.22)}$$

where L is the length of the ditch system over which the interaction takes place, and for the Field Two ditch system was 5591m.

For all fields, the hydraulic gradient (the difference between in-field water table levels and ditch water levels) varied on a roughly seasonal basis. Throughout the entire study period, ditch water levels were greater than in-field water table levels in summer, illustrated by negative gradients. During the winter months, water table levels were higher than ditch water levels (Figure 3.37). The movement of water could therefore be assumed to be from ditch to field in the summer and from field to ditch in the winter, equivalent to suggestions regarding the hydrological functioning of wet grasslands in the UK in general (Section 1.6.5). The hydraulic gradients for Fields Two and Three were similar throughout the study period (Figure 3.37). Although the relationship was characterised by a considerable degree of scatter, the slope of the relationship between the hydraulic gradient in Fields Two and Three was close to the 1:1 line (Figure 3.38.a).

Results suggested that variations in the hydraulic gradient were larger in areas where water levels were maintained for agriculture (Field One) than those on the SWT Reserve. Although this trend could not be fully substantiated due to the limited ditch water level data available for Field One, the range of hydraulic gradients evident in that field exceeded those in Field Two during the equivalent period. In Fields Two and Three the hydraulic gradient could be partially predicted using MORECS SMD data, since a logarithmic relationship between the two variables was apparent (Figure 3.39). On an annual basis, the volumes involved in the interaction between ground- and surface-water were negligible compared to other processes. On Field Two, both seepage and recharge were at least two orders of magnitude less than the volumes represented by rainfall and evaporation (Table 3.14). These results suggest that on the Pevensey Levels, and potentially in other clay-dominated areas, seepage can be omitted from water balance assessments. They also provide further support for the treatment of the phreatic and surface water components of the local hydrological cycle as two separate entities.

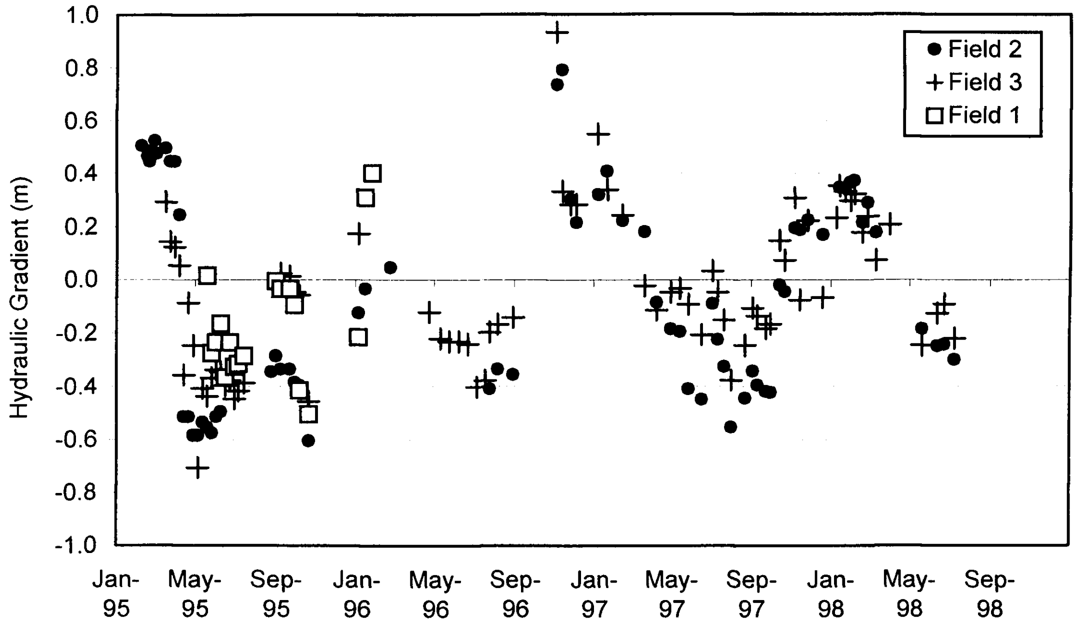


Figure 3.37. The seasonal variation of the relationship between ditch water levels and in-field water table levels (hydraulic gradient) for fields in the SWT Reserve.

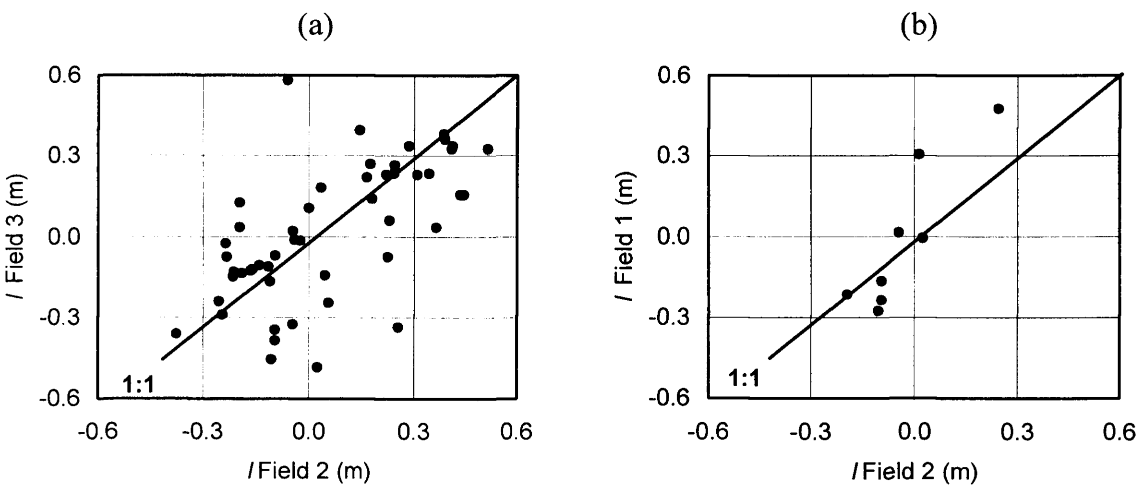


Figure 3.38. The relationship between the hydraulic gradient in Field Two and that in (a) Field Three and (b) Field One. The limited number of data points associated with (b) is a result of the decommissioning of the water level recorder in November 1996.

(a)

(b)

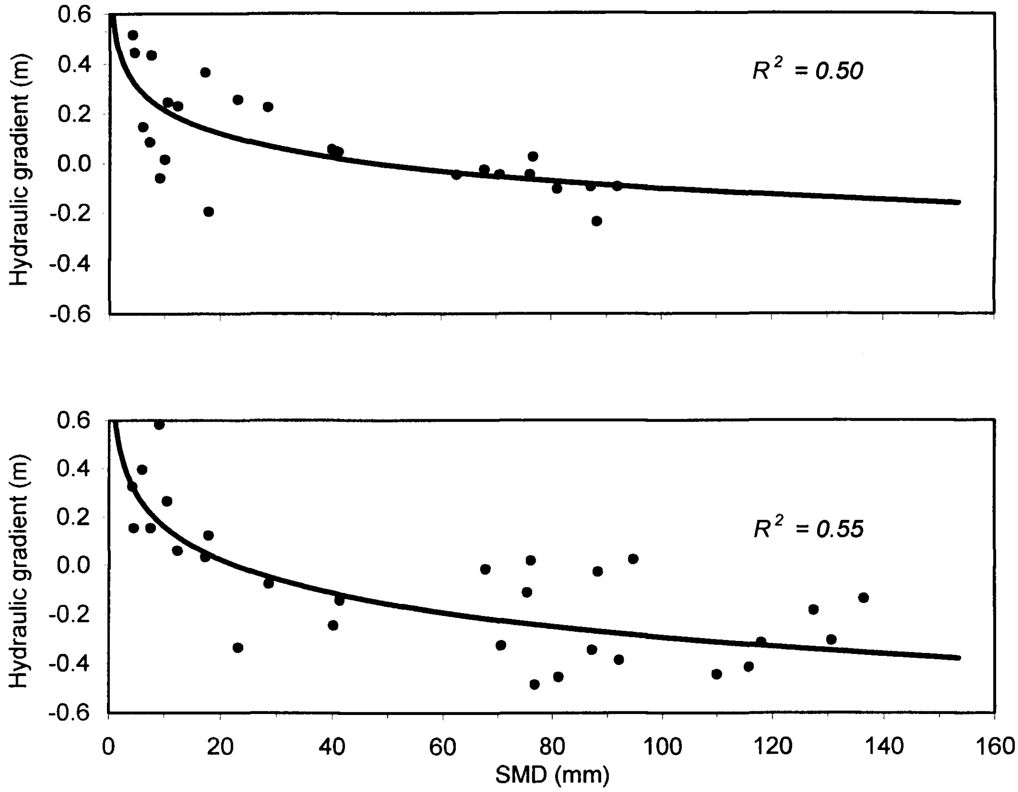


Figure 3.39. The relationship between the hydraulic gradient (*I*) (ditch water level – infield water table level) and soil moisture deficit (SMD) predicted by the Met Office MORECS system for (a) Field Two and (b) Field Three.

Year	Rainfall (m^3)	Evaporation (m^3)	Runoff (m^3)	Sluice discharge (m^3)	Ground- and surface-water interactions (m^3)
1995	12166	9605	53139	132629	6624
1996	14021	12154	43804	202348	1529
1997	21350	35247	101293	124106	3553
1998	20253	25632	88664	44513	-2431

Table 3.14. Volumes of water involved in seepage (surface to groundwater) and recharge (ground to surface water) relative to rainfall and evaporation for the Field Two ditch catchment

3.6.4. HYDROLOGY OF THE PEAT LAYER

In terms of the hydraulic gradient between field and ditch, relationships between water levels in the peat and ditch water levels were similar to those apparent in the dipwell record. In-field piezometer water levels were generally higher than ditch water levels in the winter, but lower in the summer (Figure 3.40). Data supported the model presented in Section 3.6.3 regarding the seasonal behaviour of seepage and recharge processes in wet grassland wetlands. Comparison between water levels in piezometers in the same field provided further evidence supporting this hypothesis. In winter, water levels in field-centre piezometers were generally higher than in piezometers close to the ditch with the reverse being true in the summer (Figure 3.40). Nevertheless, an important difference relative to the hydrology of clays on the wetland was that the piezometer record suggested a greater rate of water movement through the peat layer, in accordance with data describing the hydraulic conductivities of peats presented in Table 1.8.

No data regarding the hydraulic conductivities of peats on the Pevensey Levels were available. However, in Field Three, the generally high values of R^2 obtained by regression between piezometer water levels and ditch water levels (Figure 3.41) supported suggestions by Douglas and Hart (1994) regarding the potentially rapid movement of water within the peat layer and the connectivity between the peat layer and the ditches. However, a clear relationship was not so apparent for Field Two, where the relationship between ditch water level and piezometer water level was weaker (Figure 3.41). Results suggested that, in this area at least, the peat and the ditch were not connected, supporting previous suggestions regarding the discontinuous nature of the peat layer. This has been noted as a feature of soil stratigraphy on the Pevensey Levels wetland (see Section 2.3). Nevertheless, for both fields, water levels in piezometers close to the ditch (piezometer A or C) and in the field centre (piezometer B) showed close correspondence. This was taken as evidence for the existence of a hydrological equilibrium within the peat layer, even where the peat layer was not connected to the ditch. However, evidence for hysterisis in Figure 3.42.b probably indicates some form of lag in the response of field centre piezometers to inflows from the ditches. Nevertheless, in all cases the relationships between water levels in different piezometers were characterised by high values of R^2, with the slope of the regression close to the 1:1 line (Figure 3.42).

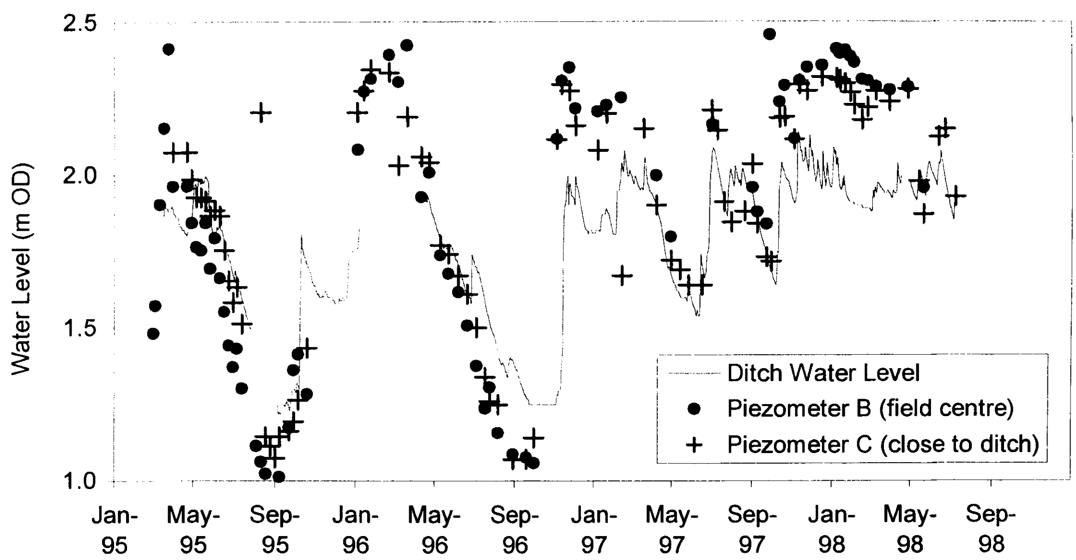


Figure 3.40. Ditch water levels and piezometer levels in Field Three 1995-1998.

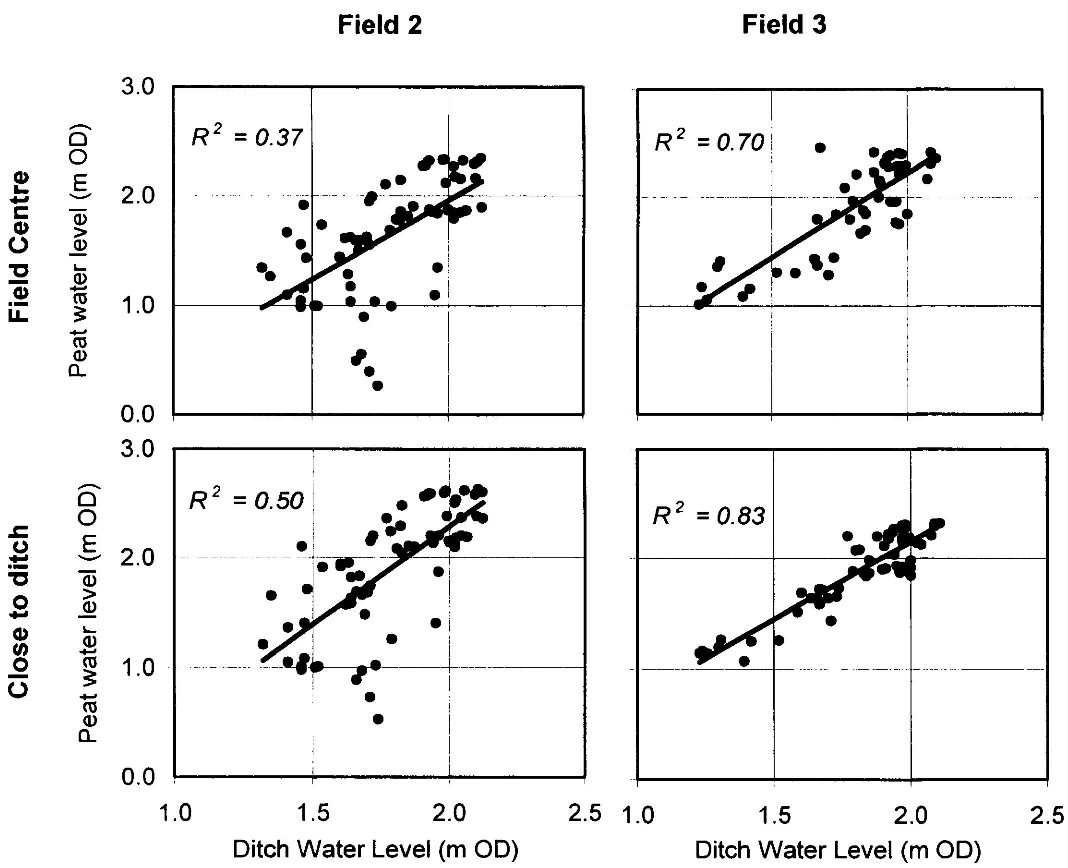
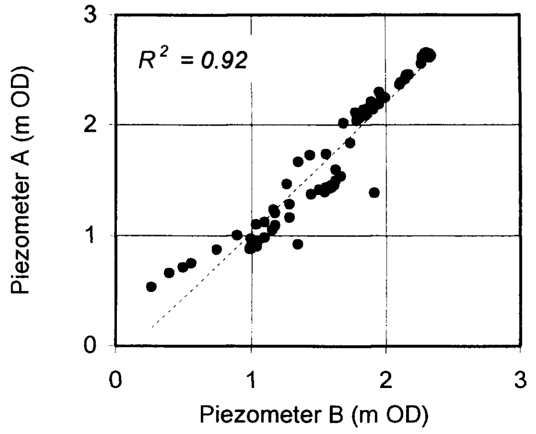
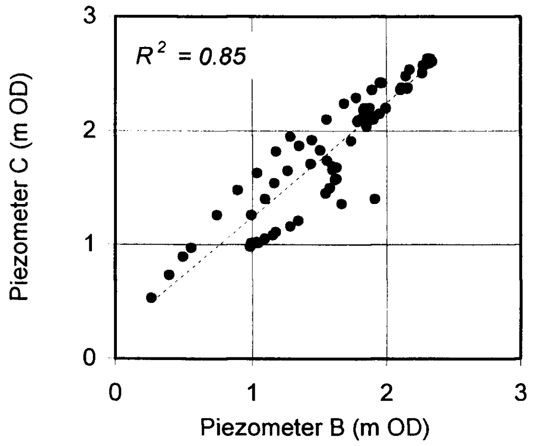


Figure 3.41. The relationship between piezometers and ditch water levels in Fields Two Three of the SWT Reserve. Based on data for the period 1995 to 1998.

(a)



(b)



(c)

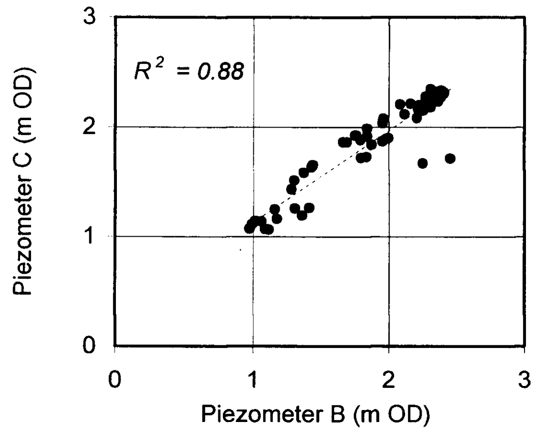


Figure 3.42. The relationship between field-centre piezometers (Piezometer B) and piezometers close to the ditch (Piezometer A or C) between 1995 and 1998 for piezometers in (a & b) Field Two and (c) Field Three.

3.6.5. STORAGE IN THE GRAVITY-DRAINED AREA

Water levels in the Manxey Haven/Old Haven, an IDB channel intersecting the gravity area were measured every time dipwells were monitored between 1995 and 1998, roughly on a fortnightly basis. These data were employed to estimate storage in the gravity-drained area, complimenting the data presented for the pump-drained lowland channel network on the wetland. The method employed to calculate surface water storage in the gravity area was analogous to that employed for pumped sub-catchments of the wetland and required the development of a level-volume relationship. As for pumped sub-catchments on the wetland, for the development of this relationship, all channels were assumed to be aligned relative to the centroid of respective cross sections and water levels in IDB channels were taken as representative of levels in Type 1 channels (Section 3.5.3). Based on these assumptions and data regarding total channel lengths in the Gravity area, the level-volume relationship was given by

$$\text{Volume} = 122688\, l + 310825 \qquad \text{(Equation 3.23)}$$

where l was water level in the Manxey Haven/Old Haven (m OD).

One important difference relative to studies of surface water storage in pumped sub-catchments was that in the gravity area, data were available that allowed the accuracy of the latter assumption to be tested. The relationship between ditch water levels in the Manxey Haven/Old Haven relative to water levels on the Nature Reserve has been previously shown in Figure 2.8. Throughout the period for which data were available, summer water levels were closely coincident. However, there was a difference of up to 0.4m between water levels in the IDB channel and the nature reserve during the winter months. This provided some indication of the likely errors associated with assuming that water levels in IDB or pumped channels are replicated elsewhere in the sub-catchment. However, in the case of the nature reserve, an important mitigating factor is that operational management actively encourages isolation form the drainage system, so that results are not necessarily representative of the wetland as a whole.

3.7. Water Balance of the Pevensey Levels 1995-1998

3.7.1. TRENDS

A monthly component water balance for the Pevensey Levels 1995-1998 is shown in Figure 3.43. For each component of the water balance, assumptions associated with their quantification are reviewed in Table 3.15. The validity of some of these assumptions are tested in later sections. Results highlight the key roles played by rainfall and evaporation in terms of overall water availability, a feature of wet grassland wetlands previously noted by Hollis and Thompson (1996), Gilman (1989; 1990) and Cook and Moorby (1993). Rainfall accounted for at least 40% of all wetland inflows in any month, although during winter this value frequently exceeded 60% (Figure 3.44.a). As a result, rainfall during the study period accounted for the largest proportion of all inflows on an annual basis (Table 3.16). Inflows from the Wallers Haven were large in winter, sometimes approximating the contributions associated with rainfall. Throughout the four-year study period, $Q_{\text{Wallers Haven}}$ was, on average, 71% of rainfall contributions. Contributions from sewage treatment works (STWs) were negligible on an annual basis (Table 3.16), although during dry summers such as those in 1995 and 1996, contributions from the STWs exceeded 10 % of all wetland inflows (Figure 3.44.a).

The largest proportion of outflows on an annual basis were those associated with evapotranspiration and evaporation (Table 3.16). During the summer months (May-September), water losses by this process represented up to 80% of all outflows from the wetland (Figure 3.44.b). Losses to sea from the wetland during winter accounted for equivalent proportions of water lost by evapotranspiration and evaporation in the summer (Figure 3.44.b), although in volumetric terms, losses to sea on an annual basis were smaller (Table 3.16). The magnitude of losses through tidal sluices highlighted the need for the continued collection of hydrological data describing the hydrology of the embanked channels on the wetland, including water levels, gate levels and volumes pumped. Calculations suggested that the assumption that all water pumped into embanked channels was lost to sea was inappropriate. During winter, losses to sea frequently exceeded lowland pumping (Figure 3.44.b) identifying the important role played by tidal sluices in evacuating winter inflows from the Wallers Haven. The smaller proportions of lowland pumping apparent during the summer months is an indication of the amounts of water pumped from the lowland ditch network that are 'recycled' and used for lowland feeding during the summer months (Figure 3.45.b).

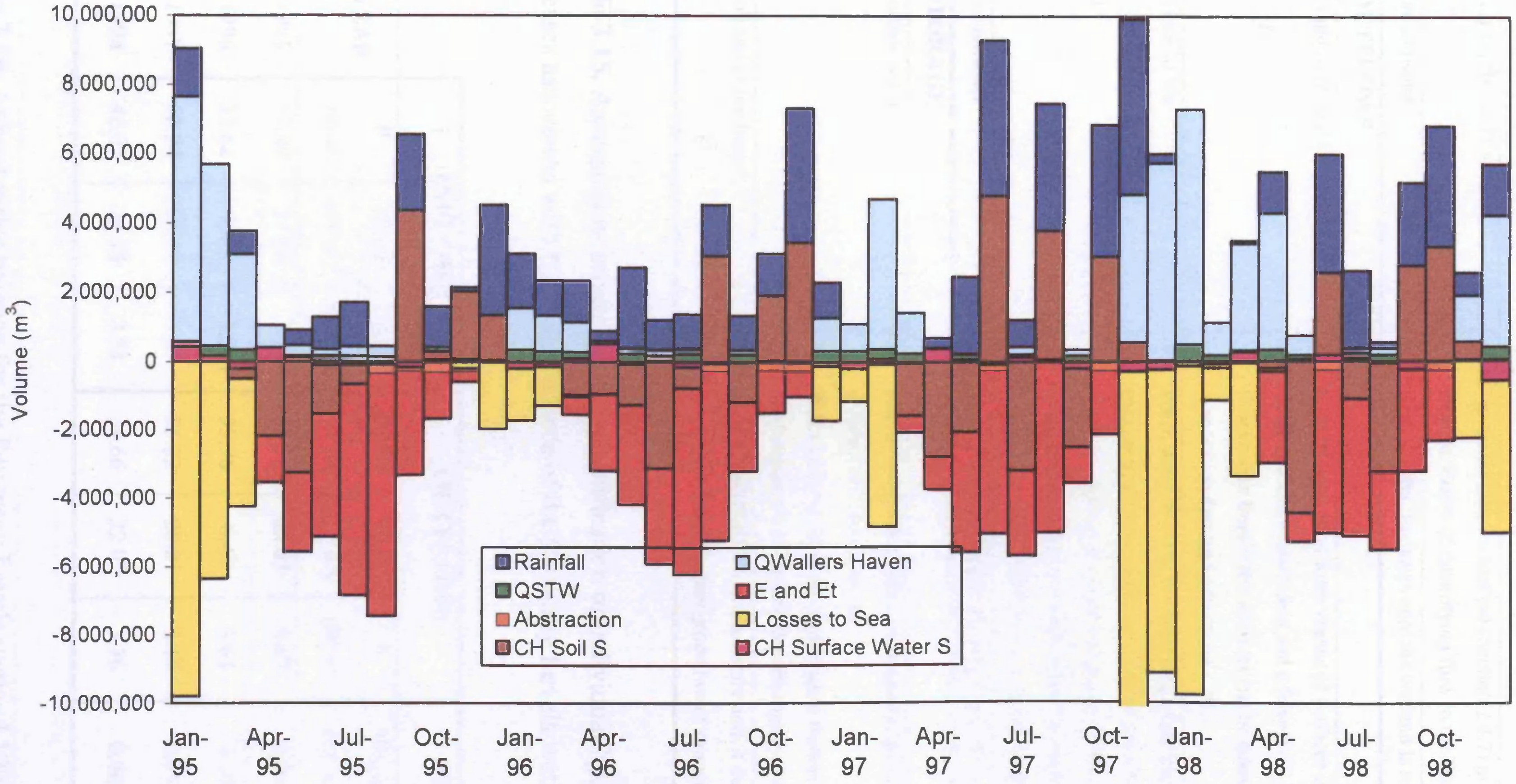


Figure 3.43. Monthly component water balance for the Pevensey Levels wetland 1995-1998.

INFLOWS	
Wallers Haven Flow	• The factor formula method (Section 2.4.7) provides an accurate means of quantifying flow in the Wallers Haven
Groundwater	• Groundwater discharge onto the wetland is negligible
OUTFLOWS	
Evaporation and Evapotranspiration	• Evapotranspiration from vegetated surfaces can be quantified using tank evaporation data and a factor of 0.88 • Evaporation from water surfaces can be quantified using tank evaporation data and a factor of 1.00
Losses to sea	• When water levels at the beginning of the month are below the gate level, all losses from the channel can be ascribed to evaporation from the water surface and lowland feeding • Pump hours provide a reliable means of calculating pumped inflows to the embanked channels on the wetland
Groundwater	• Seepage of groundwater from the wetland is negligible
STORAGE	
Surface water	• Ditches of different types are aligned relative to the centroids of respective cross-sections • Water levels at the head of embanked channels and wetland sub-catchments adequately describe upstream conditions
Soil and groundwater	• Met Office MORECS soil moisture deficit data can be used to quantify soil and shallow groundwater storage in wetlands

Table 3.15. Assumptions implicit in the quantification of individual hydrological processes associated with the water balance of the Pevensey Levels wetland.

	INFLOWS			**OUTFLOWS**			**STORAGE**	
YEAR	P (10^6 m^3)	Q_{WH} (10^6 m^3)	Q_{STW} (10^6 m^3)	E&ET (10^6 m^3)	Q_{Sea} (10^6 m^3)	A (10^6 m^3)	$\Delta S_{Surface}$ (10^6 m^3)	ΔS_{Soil} (10^6 m^3)
1995	37.90	21.59	2.93	39.56	22.63	3.67	0.00	0.74
1996	32.62	9.15	3.24	33.96	6.48	3.65	-0.05	-0.07
1997	50.24	21.17	3.25	35.82	28.01	3.17	0.05	-0.21
1998	46.02	26.15	3.58	34.66	22.02	3.36	0.00	-0.27

Table 3.16. Annual water balance for the Pevensey Levels wetland 1995-1998. See Section 3.2.1 for notation.

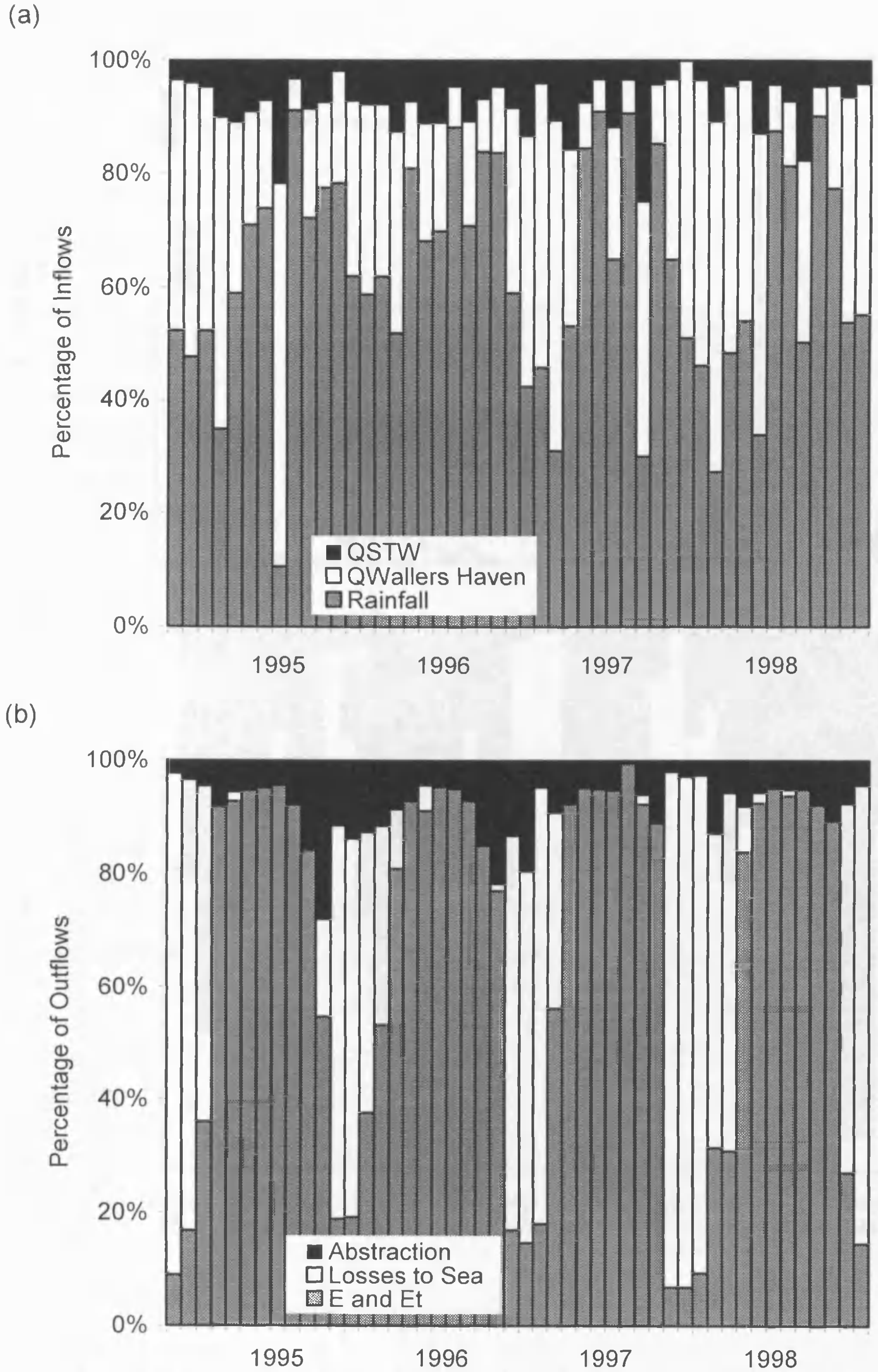


Figure 3.44. Proportion of monthly inflows (a) and outflows (b) associated with different hydrological processes on the Pevensey Levels wetland 1995-1998.

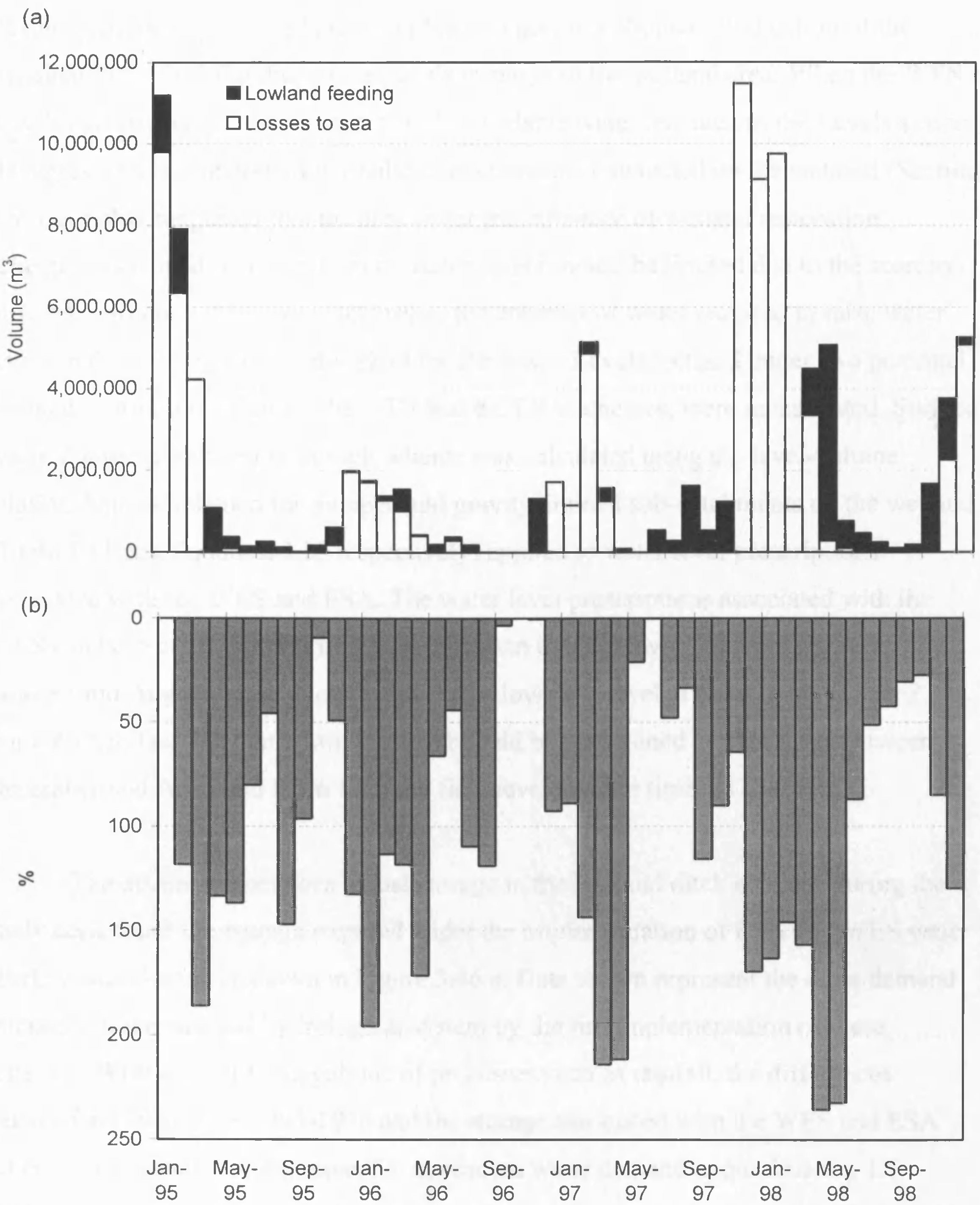


Figure 3.45. (a) Volumetric losses to sea and feeding from embanked channels on the wetland 1995-1998 and (b) the percentage of the volume pumped that feeding and losses to sea represent.

3.7.2. VIABILITY OF PROPOSED WATER LEVEL MANAGEMENT

Results provided by the wetland water balance give a preliminary indication of the sustainability of raising ditch water levels throughout the wetland area. When the WES was drawn up no account was taken of the available water resource in the Levels system (Douglas, 1993). Previous water balance assessments conducted on the wetland (Section 2.5) have also suggested that the area under the influence of wetland restoration schemes associated with higher ditch water levels should be limited due to the scarcity of water resources. To address this issue, the volumes of water required to raise water levels in the lowland ditch network of the Pevensey Levels wetland under two potential wetland restoration schemes, the WES and the ESA schemes, were investigated. Surface water storage associated with each scheme was calculated using the level-volume relationships established for pumped and gravity drained sub-catchments on the wetland (Table 3.11 and Equation 3.23 respectively) applied to water level prescriptions associated with the WES and ESA. The water level prescriptions associated with the WES can be broadly summarised as no less than 0.3m below field level between January and August, and no less than 0.6m below field level at other times (Table 2.13). For the ESA Tier 3 scheme, water levels should be maintained at field level between December and April and 0.3m from the field level at other times (Table 1.15).

The difference between actual storage in the lowland ditch network during the study period and the storage required under the implementation of ESA and WES water levels wetland-wide is shown in Figure 3.46.a. Data shown represent the extra demand imposed on the wetland hydrological system by the full implementation of these schemes. With respect to magnitude of processes such as rainfall, the differences between actual storage 1995-1998 and the storage associated with the WES and ESA schemes was small. For example, the maximum water demand approximating 1.1 million m^3 associated with the ESA scheme in the winter months (Figure 3.46.a) could be satisfied by a rainfall event of 19.3mm, assuming that 100% of the wetland catchment contributed to runoff. However, during the summer months, smaller volumes of water would generally be required since water levels on the wetland during this period are already maintained at high levels to supply drinking water for cattle, maintain wet fences and ensure the irrigation of grass pasture and arable crops.

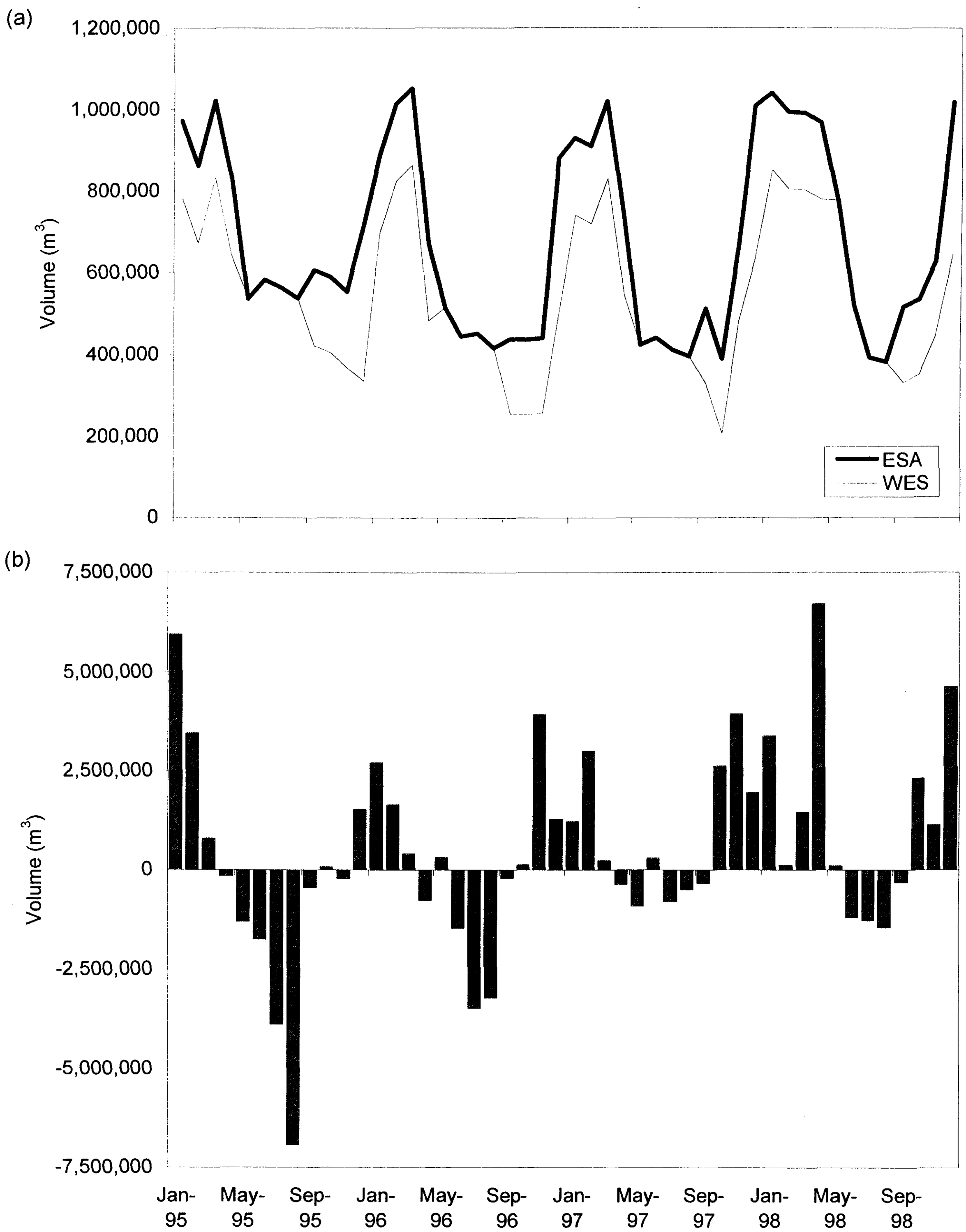


Figure 3.46. (a) Volumes of water required to raise ditch water levels wetland-wide from actual water levels to WES and ESA prescriptions. (b) shows the net water balance (the balance between wetland inflows, outflows and changes in storage) during the same period.

Water balance calculations indicate that during the winter months, the volumes of water required by both the WES and ESA schemes could be easily satisfied due to the net positive balance between wetland inflows and outflows (Figure 3.46.b). In contrast, the lower summer demand associated with the ESA and WES schemes coincides with periods when wetland outflows exceeded inflows. For all years, the period during which outflows exceeded inflows lasted between March and September (Figure 3.46.b), except for 1998 when the period in deficit began in June. The magnitude of water deficits was greatest in 1995 and 1996 and smallest in 1997 and 1998. The total water resource deficit between April and September in 1995 and 1996 was 14,5 million m^3 and 8.9 million m^3 respectively compared to 2.7 million m^3 and 4.2 million m^3 for 1997 and 1998.

Results highlight the potential difficulties of attaining WES and ESA prescriptions on the wetland during most summers, but especially during dry summers such as those of 1995 and 1996. Of particular importance was the timing of the net hydrological deficit. The start of the deficit in April coincided with the traditional timing of the reversion of wetland hydrological management to summer settings (Section 2.4.3). Results suggest that the reversion to summer conditions will have to take place earlier if additional water to supply wetland restoration strategies is to be retained. One potential option might be to reduce losses to sea during spring by implementing summer levels on the gates of embanked channels earlier in the year.

The effect on the net balance between wetland inflows and outflows of reverting to summer gate levels prior to April is shown in Figure 3.47. Closing gates in March had a negligible influence on the balance between inflows and outflows. Indeed reverting to summer settings as early as February had a limited influence on the timing and duration of the period during which outflows exceeded inflows in any year, although for the winters of 1995 and 1997 some changes in the magnitude of inflows over outflows was recorded. Results provided an indication of the over-riding importance of other wetland outflows. Data shown in Figure 3.44.b show that by March, losses from the wetland by evaporation and evapotranspiration already account for 40% of wetland outflows in most years. The limited influence on the timing, duration and magnitude of water resource deficits on the Pevensey Levels of altering the management of the gates on embanked channels is ascribed to the large influence evaporative losses have on the hydrology of the wetland.

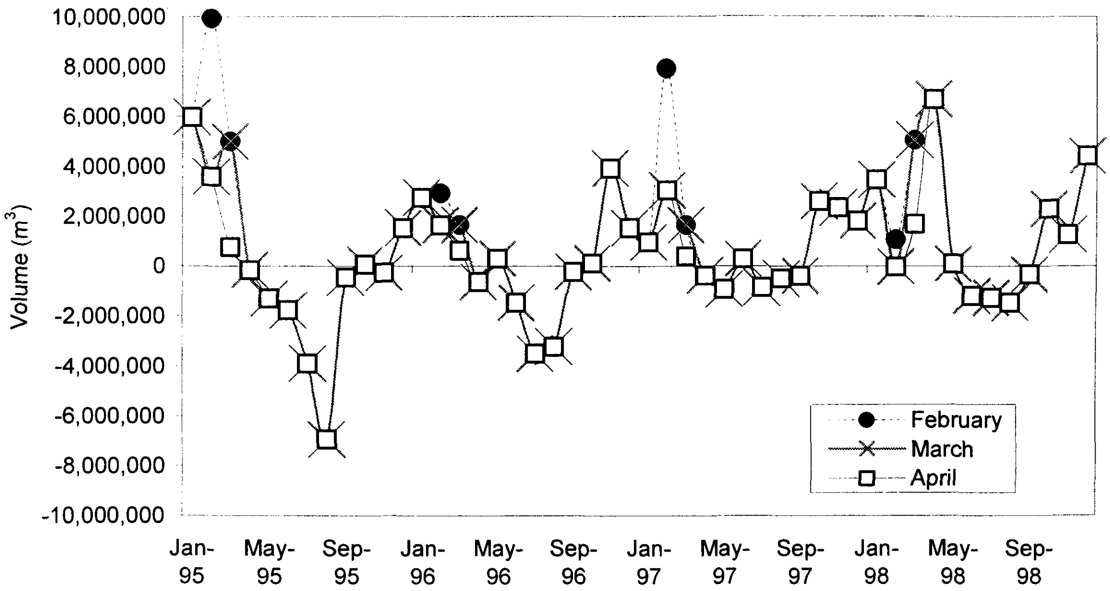


Figure 3.47. Comparison of the balance between wetland inflows and outflows on the Pevensey Levels wetland 1995-1998 due to reverting to summer gate settings on embanked channels at different times of year.

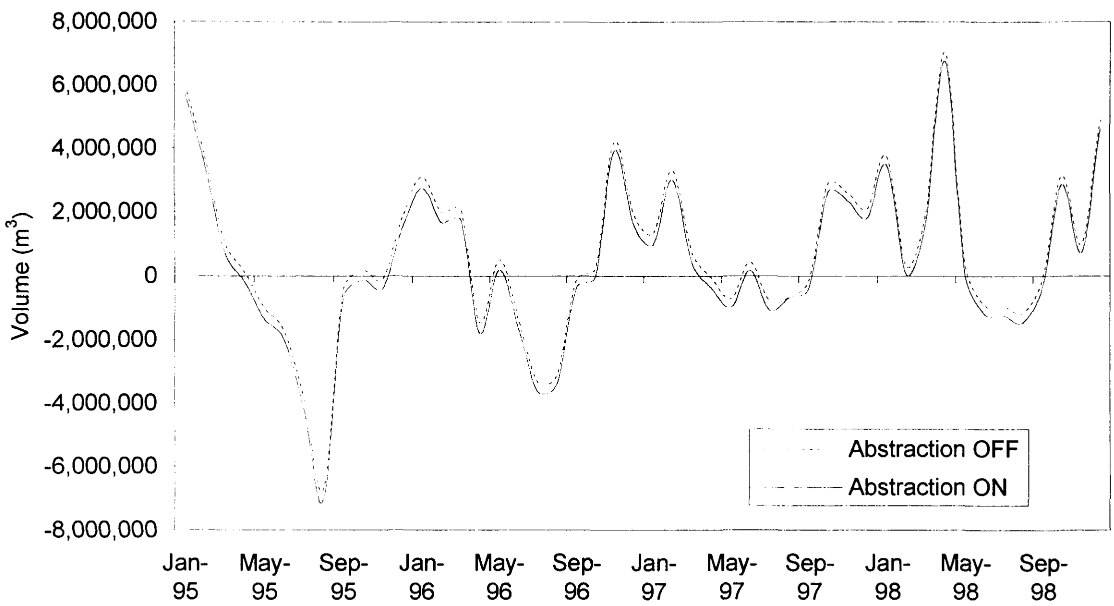


Figure 3.48. The effects of abstraction on the net water balance of the Pevensey Levels wetland.

3.7.3. MANAGING THE WALLERS HAVEN WATER RESOURCE

Results provided by the wetland water balance also provide an indication of the potential implications of abstraction and revised water level management to sustainable water resource management in the Wallers Haven. The volumes of water abstracted for public water supply were small when considered relative to losses to sea or evapotranspiration (Figure 3.43). Abstraction rarely exceeded 15 % of all outflows from the wetland (Figure 3.44.b). Indeed, removing abstraction from the wetland water balance had a limited influence on the balance between inflows and outflows during the study period (Figure 3.48). However, some of the largest rates of abstraction (Figure 3.49.a) coincided with the period of net water resource deficit (Figure 3.49.b), although the augmentation of upland tributaries of the Wallers Haven was a feature of all the periods for which water resource deficits were recorded (Figure 3.49.b). However, augmentation is already accounted for in the wetland water balance because augmentation boreholes are located upstream of the gauging stations used to compute Wallers Haven flow (Figure 2.5).

Based on the actual water levels maintained during the study period, meeting WES water level prescriptions in the four sub-catchments directly connected to the Wallers Haven (Star Inn, Manxey, Waterlot and Gravity sub-catchments) would require up to 0.5 million m^3 depending on the time of year. As in the case of catchment-scale calculations, the volumes of water required in the summer were generally smaller than those in the winter (Figure 3.46.a) as, during the summer months, ditch water levels are already kept high. Nevertheless, the provision of this extra storage would require the Wallers Haven to be re-profiled. Figure 3.50 shows the water levels that would ensure the provision of the additional volumes of water required by the implementation of WES in the gravity, Manxey, Waterlot and Star Inn catchments 1995-1998 relative to actual water levels during that period. Water levels have been estimated by assuming that all extra water would have to be stored above a level of 1.75m OD, since this is the minimum water level in the Wallers Haven required to feed lowland ditches (Section 2.4.5). During most of the year, storage of the extra water required for the WES and ESA schemes in Wallers Haven sub-catchments the would result in water levels in excess of the mean bank level (3.00m OD; Section 2.4.5)(Figure 3.50). Only in November-December 1995, September-November 1996, July-October 1997 and July-November 1998 would there be sufficient storage to attain WES prescriptions in the four adjoining catchments without increasing flood risk to bankside land.

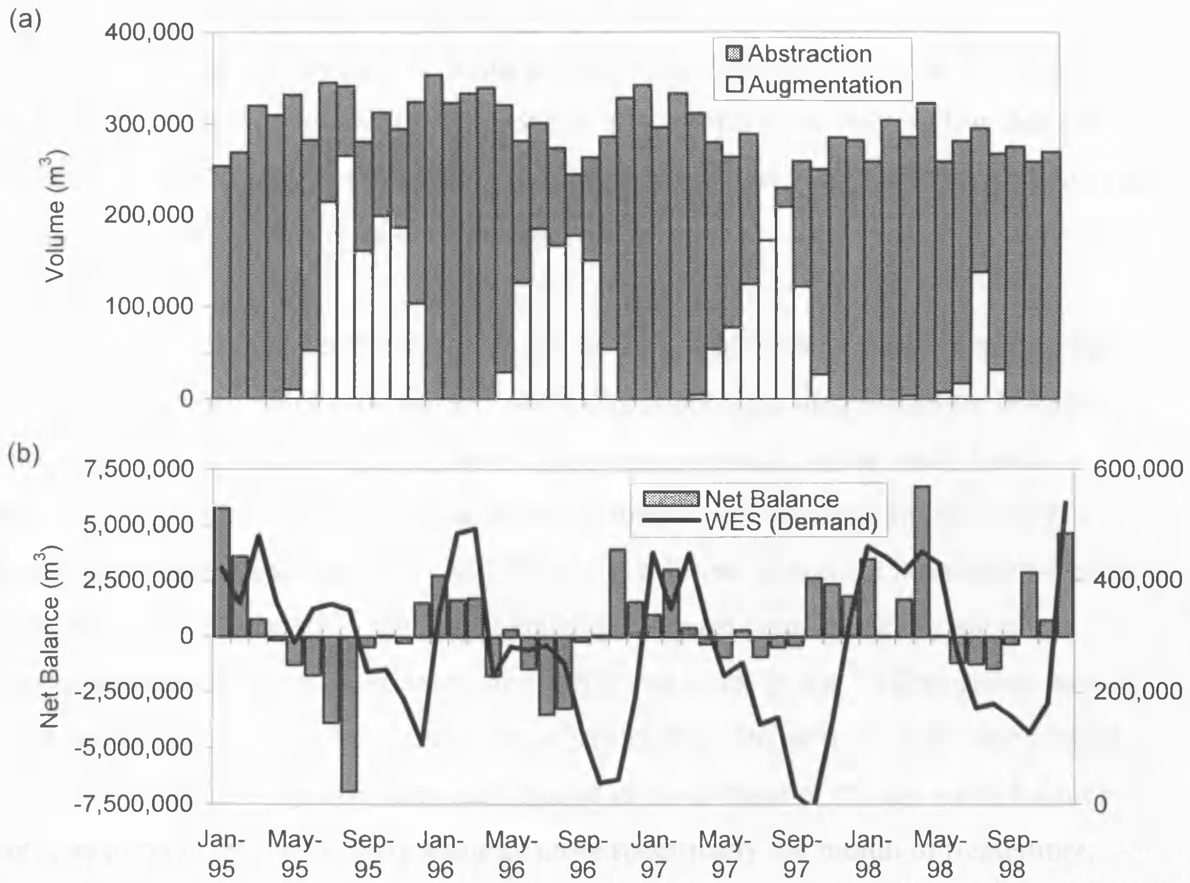


Figure 3.49. (a) Monthly abstraction from the Wallers Haven relative to (b) the net water balance, and water demand associated with the implementation of the WES in the Gravity, Star Inn and Waterlot sub-catchments 1995-1998.

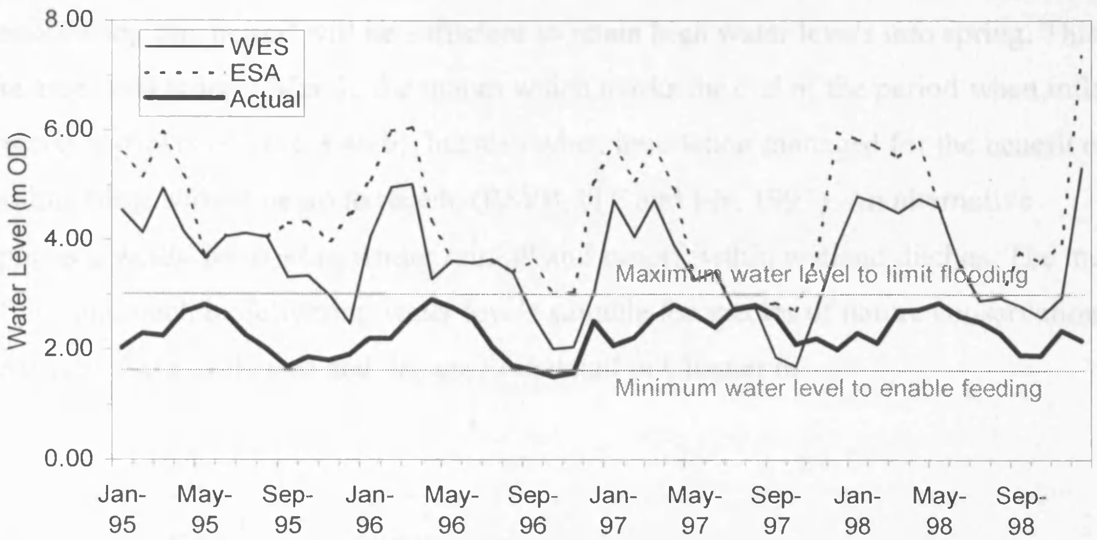


Figure 3.50. Water levels in the Wallers Haven 1995-1998 relative to the water levels required to deliver the volumes of water required to achieve WES and ESA prescriptions in the Gravity, Star Inn and Waterlot sub-catchments.

To attain ESA prescriptions, only during July-October 1997 and July-August 1998 could sufficient water be stored to attain prescriptions without the risk of flooding (Figure 3.50). During 1995 and 1996 water levels wetland-wide were so low that the amount of water required to raise them to ESA prescriptions would not be satisfied even by maximum storage in the Wallers Haven.

Results clearly identify the difficulty of satisfying water level targets for nature conservation in dry years such as 1995 and 1996, supporting suggestions by Douglas (1993; Section 2.5) regarding the need to limit the area under higher water levels. Whilst higher water levels than those currently maintained can be promoted, only during wetter years such as 1997 and 1998 can conditions of surface inundation such as those advocated by the ESA scheme be satisfied. In most years, the early autumn months offer the greatest potential to store sufficient water in the Wallers Haven to raise water levels in adjacent catchments to those associated with nature conservation based objectives. This finding is of particular interest in the context of the net water balance since, as previously stated, early autumn, more specifically the month of September, generally coincides with the end of the period of net water resource deficit (Section 3.7.2). As such, if excess inflows during this period were stored rather than discharged to sea they could be used to raise wetland-wide water levels significantly. Given the limited volumetric importance of evaporation and evapotranspiration during the autumn and winter months (Table 3.2), it is envisaged that the provision of water to lowland areas during this period will be sufficient to retain high water levels into spring. This is the case until at least March, the month which marks the end of the period when inflows exceed outflows (Figure 3.46.b), but also when inundation managed for the benefit of wading birds should begin to recede (RSPB, ITE and EN, 1997). An alternative approach would be to store winter rainfall and runoff within wetland ditches. The merits of this approach in delivering water levels suitable for species of nature conservation importance are evaluated and discussed in detail in Chapter 6.

3.7.4. ACCURACY OF THE WETLAND WATER BALANCE METHOD

In previous sections, error has been described as the residual of the wetland water balance, calculated using an inverse form of Equation 3.1, where:

$$\text{Error} = \text{Inflows} - \text{Outflows} \; +/- \; \Delta S \qquad \text{(Equation 3.24).}$$

Figure 3.46.b has employed water balance residuals as a means of evaluating wetland water availability for wetland restoration schemes. In general, negative errors were evident during summer months and positive errors during winter (Figure 3.46.b). Whilst these data clearly provide an indication of periods of water surplus at times of water scarcity, they must necessarily be considered in the context of the accuracy of the input data that are used to quantify the individual components of the water balance. Given the errors inherent in most measuring devices, deviations from zero can be expected (Rushton, 1996). Such errors will be expected from raingauges and evaporation tanks, where siting and maintenance are important for data quality (Shaw, 1993). This has important implications for calculations in wetlands such as the Pevensey Levels, where rainfall and evaporation are the dominant processes effecting wetland inflows and outflows (Section 3.7.1; Table 3.16). The assumptions adopted to quantify individual processes also introduce uncertainty to the water balance calculation. Assumptions employed for the water balance calculation of the Pevensey Levels wetland have been previously reviewed in Table 3.15.

On the Pevensey Levels, most of the assumptions could be tested due to the large volume of data describing many of the water balance components. For example, Figure 3.30 has indicated that the volumes of water pumped into embanked channels may be over-estimated by application of the method proposed by Marshall (1989; see Equation 3.12), especially where axial-type pumps are present. In the context of the hydrology of embanked channels, a further potential error associated with calculations is the use of monthly time-series for the management of retention gates located at the head of each system. This approach does not account for responses by flood defence officials to extreme rainfall events occurring over daily, as opposed to monthly, temporal scales. However, the limited sensitivity of the water balance residual to varying pumping and losses to sea, except during wet winters such as 1997 and 1998, (Figure 3.51) indicates that the assumptions made with regards to both processes are adequate for catchment-scale water balance studies.

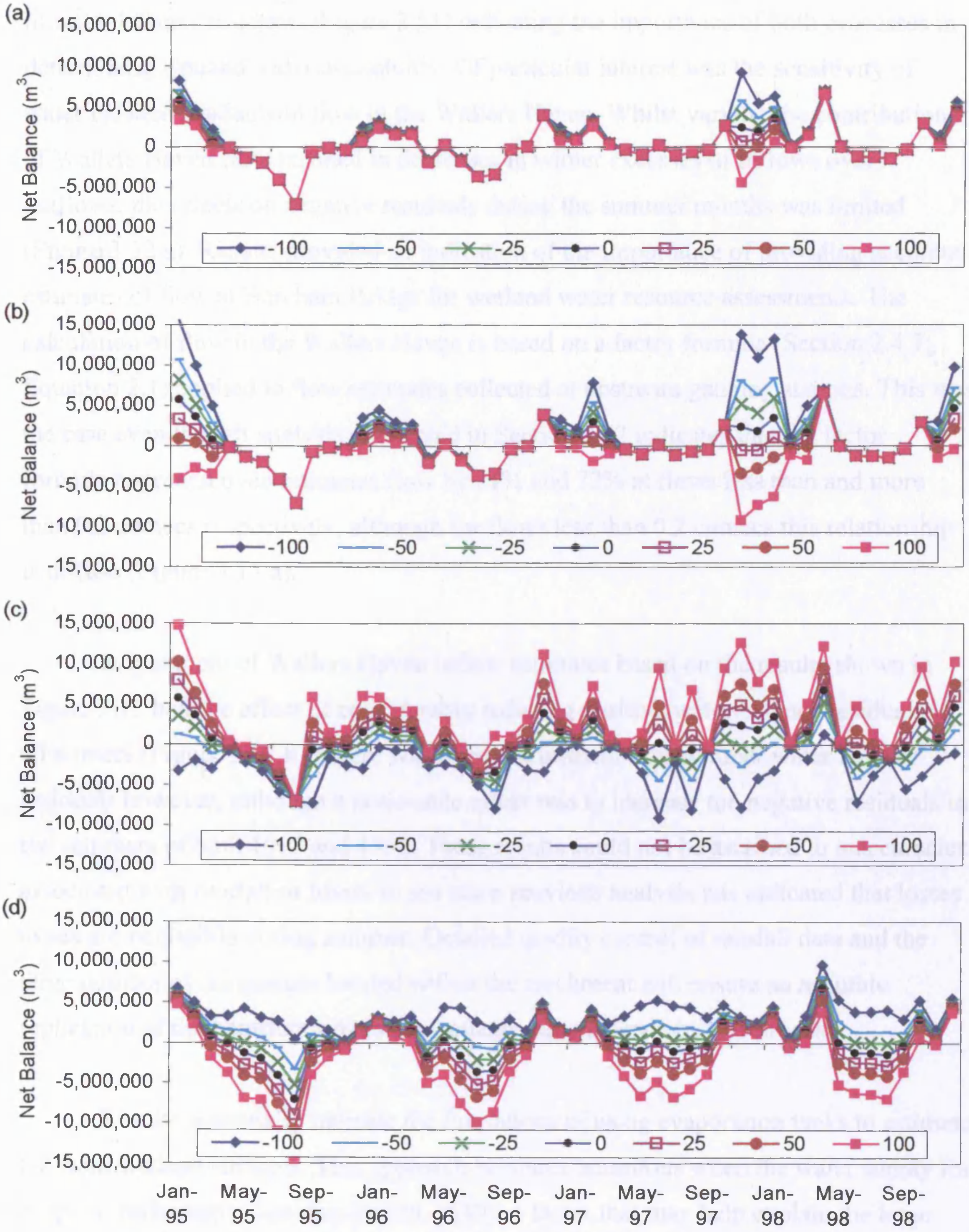


Figure 3.51. Sensitivity of monthly water balance residuals 1995-1998 to changes in (a) pumping, (b) losses to sea, (c) rainfall and (d) evaporation and evapotranspiration.

The water balance model was most sensitive to changes to rainfall and evaporation/evapotranspiration. Small changes to either parameter caused large changes in water balance residuals (Figure 3.51) indicating the importance of both processes in determining wetland water availability. Of particular interest was the sensitivity of water balance residuals to flow in the Wallers Haven. Whilst varying the contributions of Wallers Haven flow resulted in decreases in winter excesses of inflows over outflows, the effects on negative residuals during the summer months was limited (Figure 3.52.a). Results provided an indication of the importance of providing accurate estimates of flow at Boreham Bridge for wetland water resource assessments. The calculation of flow in the Wallers Haven is based on a factor formula (Section 2.4.7; Equation 2.1) applied to flow estimates collected at upstream gauging stations. This was the case even though analysis conducted in Section 3.42 indicates that the factor formula approach over-estimates flow by 21% and 72% at flows less than and more than 0.2 cumecs respectively, although for flows less than 0.2 cumecs this relationship is diffuse (Figure 3.17.a).

Adjustment of Wallers Haven inflow estimates based on the results shown in Figure 3.17 had the effect of considerably reducing positive water balance residuals in all winters (Figure 3.52.b). There was a limited influence on summer water balance residuals however, although a noticeable effect was to increase the negative residuals in the summers of both 1997 and 1998. These results could not be ascribed to inaccuracies associated with rainfall or losses to sea since previous analysis has indicated that losses to sea are negligible during summer. Detailed quality control of rainfall data and the large number of raingauges located within the catchment will ensure an accurate replication of the contributions to the wetland water balance by this process.

Results potentially indicate the limitations of using evaporation tanks to estimate ET from wetland surfaces. This approach becomes hazardous when the water supply for evaporation becomes limiting (Smith, 1992), a factor that may help explain the large negative residuals apparent during the summers of 1995 and 1996. Similarly, positive residuals in the wetter-than-average summers of 1997 and 1998 could be ascribed to the fact that in wetlands, where the water is close to the surface, evaporation rates may proceed at rates greater than those predicted by traditional evaporation estimation techniques (Crundwell, 1987). The dynamics of evapotranspiration and the influence of different estimates on the wetland water balance is considered in the following Chapter.

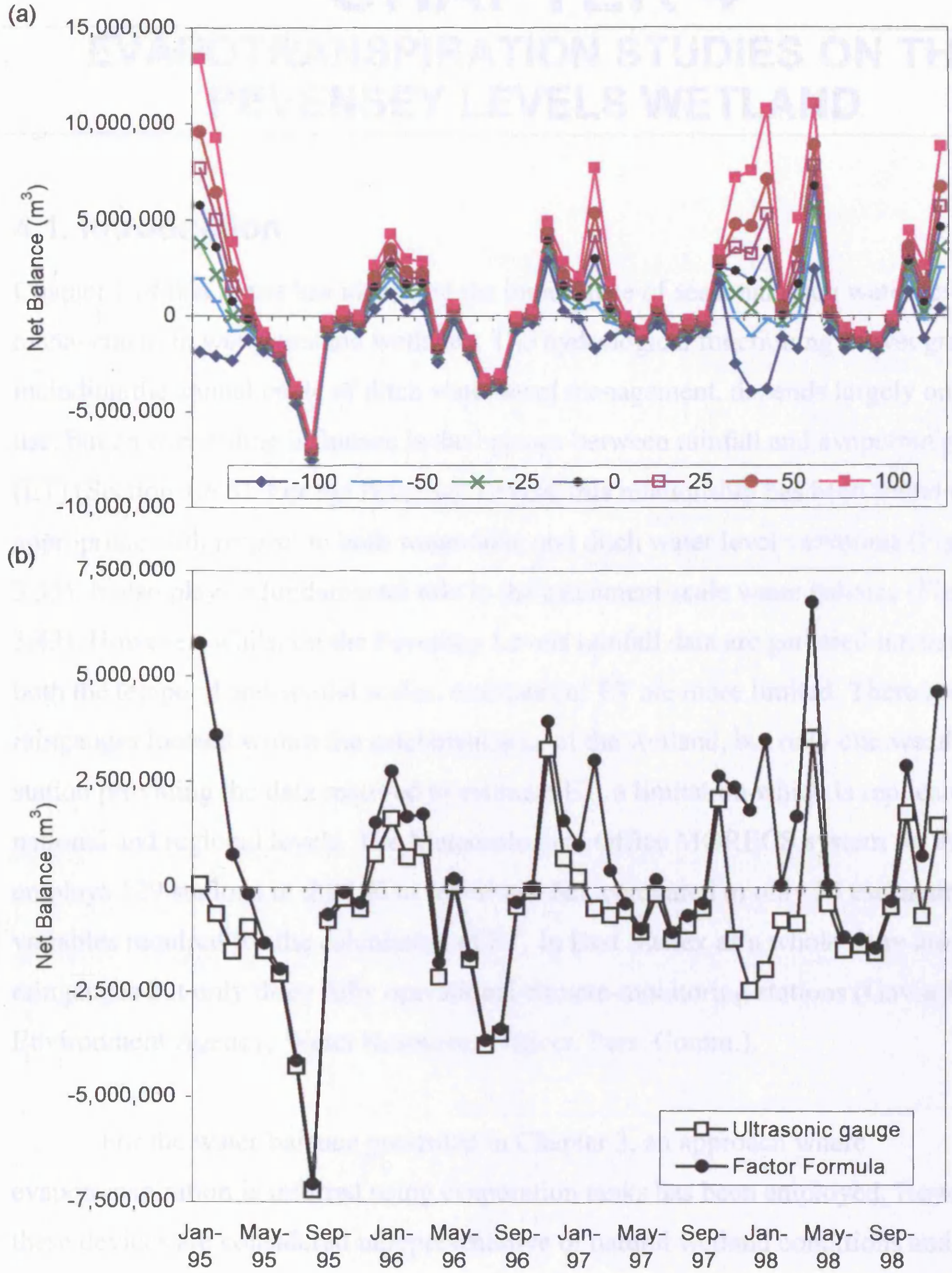


Figure 3.52. (a) Sensitivity of water balance residuals 1995-1998 to variations in the contributions of upland tributaries of the Wallers Haven and (b) the influence on water balance residuals of adjusting Wallers Haven flow estimates based on flow estimates provided by the ultrasonic flow gauge (see Section 3.4.2).

CHAPTER 4

EVAPOTRANSPIRATION STUDIES ON THE PEVENSEY LEVELS WETLAND

4.1. Introduction

Chapter 1 of this thesis has identified the importance of seasonal ditch water level management in wet grassland wetlands. The hydrological functioning of wet grassland, including the annual cycle of ditch water level management, depends largely on land use, but an over-riding influence is the balance between rainfall and evapotranspiration (ET) (Section 1.6.3). For the Pevensey Levels, this relationship has been found to be appropriate with respect to both water table and ditch water level variations (Figure 3.33). It also plays a fundamental role in the catchment-scale water balance (Figure 3.43). However, whilst on the Pevensey Levels rainfall data are gathered intensively at both the temporal and spatial scales, estimates of ET are more limited. There are eight raingauges located within the catchment area of the wetland, but only one weather station providing the data required to estimate ET, a limitation which is replicated at the national and regional levels. The Meteorological Office MORECS system for example, employs 129 stations in the UK to report rainfall, compared to only 55 estimating the variables required for the calculation of ET. In East Sussex as a whole there are 64 raingauges but only three fully operational climate-monitoring stations (Gavin Johnson, Environment Agency, Water Resources Officer, Pers. Comm.).

For the water balance presented in Chapter 3, an approach where evapotranspiration is inferred using evaporation tanks has been employed. However, these devices are considered unrepresentative of natural wetland conditions and are prone to overestimation (van Keulen and Wolf, 1986, American Society of Civil Engineers [ASCE], 1996), mainly because they have a small surface area, and are liable to heat advection and exchange through the side walls (Lansley, 1998). A common approach is therefore to adjust tank values to simulate ET from different vegetation and land cover types. Factors ranging from 0.54 to 5.3 of tank ET are reported in a review by Carter *et al.* (1979). In Chapter 3, a factor of 0.88 Eo_{Tank} has been used to replicate the methods used for the estimation of ET wetland losses in previous water balance calculations (Section 2.5).

Some indication of the errors associated with using a time-invariant coefficient for the estimation of ET have been provided in Chapter Three relative to the residual term in water balance calculations (Section 3.7.4). Water balance data suggest that losses from the wetland by ET are over-represented, especially during dry summers. This chapter therefore considers the adequacy of methods currently employed on the Pevensey Levels wetland for the estimation of ET. An initial assessment has been conducted in Section 3.3.2.3, but the study presented in this chapter is concerned primarily with the variation of ET over shorter temporal scales. The adequacy of existing estimates is considered by comparing rates of ET inferred from climatic data to rates measured by a state-of-the-art micro-meteorological device. In later sections, results are interpreted to assess the accuracy and validity of current estimates of ET incorporated within water balance calculations. Subsequent chapters incorporate findings within more temporally-intensive modelling studies of the hydrological functioning of the Pevensey Levels wetland.

4.2. Measurement and estimation of ET

The most significant gap in the knowledge of wetland systems is the lack of detailed information on the processes affecting water levels, particularly rates of evapotranspiration (Duever, 1988). As a result, in modelling wetland hydrology, ET is normally considered as a function of tank evaporation or the potential rate of ET. In combination, the requirement of high quality instrumentation, prohibitive cost (Table 4.1) and the need for routine maintenance by an experienced operator (Lansley, 1998) make the direct measurement of ET unsuitable for long term monitoring (Souch *et al.*, 1996). A more common approach is therefore to estimate ET indirectly.

The simplest way to estimate ET is by subtracting annual runoff from annual precipitation (Claasen and Halm, 1996). Other indirect methods include the estimation of ET as a residual of the water balance (Yin and Brook, 1992) and the use of lysimeters (Gilman, 1994). However, field water balances demand considerable instrumentation at both upper and lower boundaries of the soil-plant-atmosphere system under study (Villagra *et al.*, 1995) and their applicability is therefore limited in wetlands where hydrological knowledge is generally lacking, an aspect that characterises wet grassland areas (Cook and Moorby, 1993). The application of this approach is especially problematic where not all the variables involved in the wetland water balance have been quantified. In these areas, the calculation of AET as a residual of the water balance may involve the adoption of unrealistic assumptions regarding the importance of other hydrological processes. In wetlands, the interaction between ground- and surface water can be important in volumetric terms but, in common with evapotranspiration, is difficult to measure directly (Meyboom, 1966; Carter and Novitski, 1988; Said, 1993). Lysimeters have been widely employed for the estimation of ET, but differences between inside and outside conditions, including soil characteristics and moisture availability may lead to problems of representativity (ASCE, 1996). As a result, the consensus is that it is probably difficult to obtain better than plus or minus 40% accuracy in wetland evapotranspiration estimates (Ingram, 1983).

Because of the limitations of indirect methods, ET is most commonly estimated by application of routinely made measurements of meteorological variables to empirical and semi-empirical equations (WMO, 1994). There are a hierarchy of equations which express the transfer of water vapour between the surface and the atmosphere (Stewart, 1989). The most commonly quoted methods are reviewed in Table 4.2. Shuttleworth (1979) provides a detailed review of these methods. A shared aspect of all empirical and semi-empirical methods however is that they provide a 'standard,' or potential, rate of ET (Shuttleworth, 1979). Historically in the UK, the most common approach is that proposed by Penman (1948), where atmospheric measurements of net radiation, temperature, windspeed and humidity are used to estimate ET *'from a short green crop, actively growing, completely shading the ground, of uniform height and not short of water'*.

By assuming that the crop is 'not short of water', this method is concerned with ET when the only control is atmospheric demand (Loomis and Connor, 1992). However, the applicability of potential rates becomes hazardous when the vegetation experiences water stress, since the rate of ET is influenced by the restricted supply of water from the soil (Wallace, 1991). An essential distinction is therefore between the evapotranspiration that actually takes place from a vegetated surface, the Actual Evapotranspiration (AET), and the potential rate that would occur under well watered conditions (the Potential Evapotranspiration, or PET) (Ward and Robinson, 1989). For practical applications it is AET which is most often required, although the concept of PET can be used as a scale upon which the influence of surface control can be superimposed, often as a multiplication factor (Shuttleworth, 1979).

Instrument	Manufacturer	Price (£)	Notes
Bowen ratio	Campbell Scientific	5,000	Sensors only. Does not include data logger
Hydra	Centre for Ecology and Hydrology	50,000	Not manufactured commercially
Solent	Solent Scientific	12,000	Excludes net radiometer

Table 4.1. Approximate costs of instrumentation for the direct measurement of ET.

Method	Equation	Notes
Penman (1948)	$\frac{(\Delta/\gamma\ H + E_a)}{(\Delta/\gamma + 1)}$	See Section 3.3.2.1. for notation applicable to the Pevensey Levels
Makkink (1956)	$C\,(W\,R_s)$	See Appendix 4.1.for notation
Priestley-Taylor (1972)	$a.\ [\Delta/\ (\Delta/\ \gamma)]\ (R_n + G)$	See Appendix 4.1.for notation
Blaney Criddle (1950)	$C\,[P\,(0.46T + 8)]$	See Appendix 4.1.for notation
Penman-Monteith (1965)	$\frac{(\Delta\ (R_n - G) + \rho C_p\ (e_s\text{-}e)\ /\ r_a)}{(\Delta + \gamma\ (1\ + r_s/\ r_a))}$	See Section 4.4. for notation

Table 4.2. Empirical models commonly employed for the calculation of ET.

4.3. The crop coefficient approach

The crop coefficient approach is the most common method used to estimate AET from PET estimates. In this approach, estimates of AET are obtained by moderating a reference rate of PET, or PET_{Ref}, according to vegetation and soil moisture characteristics (Granger and Gray, 1989). This is the standard approach proposed by the Food and Agricultural Organisation (FAO) of the UN (Smith, 1992), and takes a two-step process. Firstly, crop potential evapotranspiration (PET_{Crop}) is calculated from PET_{Ref} by

$$PET_{Crop} = PET_{Ref} \times Kc_{\text{Crop Type}} \qquad \text{(Equation 4.1)}$$

where PET_{Ref} is defined as *'the rate of evapotranspiration from an extended surface of 0.08 – 0.15 m height of green grass cover of uniform height actively growing and completely shading the grass'* (Doorenboos and Pruitt, 1977), equivalent to Penman's idealised evaporative surface. $Kc_{\text{Crop Type}}$ is a coefficient specific to crop type, termed a crop coefficient, and accounts for differences between the stomatal characteristics of different crops relative to grass. PET_{Ref} can be obtained by the Penman, Blaney-Criddle, Makkink (Table 4.2.), or tank evaporation methods (ASCE, 1996), with values of $Kc_{\text{Crop Type}}$ varied throughout the year to account for different crop growth stages. The second step of the FAO approach involves the calculation of the rate of AET by adjusting PEt_{Crop} estimates according to soil moisture characteristics (Smith, 1992) by

$$AET = PET_{Crop} \times Kc_{\text{Water Availability}} \qquad \text{(Equation 4.2)}$$

The two step FAO model has been successfully and widely applied due to the transferability of Kc curves, its ease of application and because it gives the individual making calculations a visual representation of the process (American Society of Civil Engineers [ASCE], 1996). Considerable work has been done on measuring $Kc_{\text{Crop Type}}$ as a function of time for different crops (Stewart, 1989). For wetland and agricultural land cover types on the Pevensey Levels, appropriate values of $Kc_{\text{Crop Type}}$ for application with estimates of PET_{Ref} are summarised in Table 4.3. Doorenboos and Pruitt (1984) provide an extensive review of data specific to other crop types.

Vegetation type	Crop Coefficient		Notes
Grass and Pasture			
Grass pasture (rotation)[1]	$Kc_{Initial}$	0.40	
	$Kc_{Mid\text{-}season}$	0.85	
	$Kc_{Maturity}$	0.85	
Grass pasture (poorly managed)[1]	$Kc_{Initial}$	0.30	
	$Kc_{Mid\text{-}season}$	0.75	
	$Kc_{Maturity}$	0.75	
Grass pasture (mowed)[1]	$Kc_{Initial}$	0.95	Applicable to *Lolium* spp.and
	$Kc_{Mid\text{-}season}$	0.95	*Festuca* spp. of 0.06-0.08m height
	$Kc_{Maturity}$	0.95	
Short Vegetation (0.3m) [1]	$Kc_{Initial}$	1.05	
	$Kc_{Mid\text{-}season}$	1.10	
	$Kc_{Maturity}$	1.10	
Grass (for hay)[2]	Kc_{Mean}	0.80	Assumes excellent plant
	$Kc_{Maximum}$	1.05	population density, high fertility
	$Kc_{Minimum}$	0.60	and good irrigation
Pasture[2]	Kc_{Mean}	0.95	Assumes excellent plant
	$Kc_{Maximum}$	1.05	population density, high fertility
	$Kc_{Minimum}$	0.55	and good irrigation
Wetland vegetation			
Cattails and bulrushes [1]	$Kc_{Initial}$	0.60	Maximum crop height 2m
	$Kc_{Mid\text{-}season}$	1.20	
	$Kc_{Maturity}$	0.60	
Reeds, standing water [1]	$Kc_{Initial}$	0.80	Maximum crop height 1m
	$Kc_{Mid\text{-}season}$	0.90	
	$Kc_{Maturity}$	0.90	
Reeds, moist soil[1]	$Kc_{Initial}$	0.60	Maximum crop height 1m
	$Kc_{Mid\text{-}season}$	0.70	
	$Kc_{Maturity}$	0.70	
Reed swamp (standing water) [2]	Kc_{Mean}	0.85	
Reed swamp (moist soil) [2]	Kc_{Mean}	0.65	
Submerged vegetation[2]	Kc_{Mean}	1.10	
Floating vegetation (duckweed) [2]	Kc_{Mean}	1.05	
Flat Leaf vegetation (lillies) [2]	Kc_{Mean}	1.05	
Protruding vegetation (water hyacinth) [2]	Kc_{Mean}	1.10	
Open water[2]	Kc_{Mean}	1.10	Applicable to tank data only
Arable crops			
Winter Wheat[1]	$Kc_{Initial}$	0.30	
	$Kc_{Mid\text{-}season}$	1.15	
	$Kc_{Maturity}$	0.25	
Maize[1]	$Kc_{Initial}$	0.20	
	$Kc_{Mid\text{-}season}$	1.20	
	$Kc_{Maturity}$	0.25	
Barley, wheat, oats[1]	As for winter wheat		
Corn[1]	$Kc_{Initial}$	0.40	$Kc_{Maturity}$ is for harvest after
	$Kc_{Mid\text{-}season}$	1.15	complete field drying of grain.
	$Kc_{Maturity}$	0.55	

Table 4.3. Crop coefficients for land cover types on the Pevensey Levels wetland. Data refer to areas with a sub-humid climate (Relative Humidity ~ 45%) with moderate windspeeds ($2ms^{-1}$) (from [1]ASCE, 1996 and [2]Doorenboos and Pruitt, 1977).

Three values of $Kc_{\text{Crop Type}}$ are required to construct an FAO crop coefficient curve such as that shown in Figure 4.1. These are the Kc of the initial period (Kc_{Initial}), the Kc of the mid-season ($Kc_{\text{Mid-season}}$) and the Kc at the time of harvest, or the end of maturity (Kc_{Maturity}) (Doorenboos and Pruitt, 1977). Specific definitions associated with these Kc values are given in Table 4.4. Values of Kc_{Initial}, $Kc_{\text{Mid-season}}$, and Kc_{Maturity} are then applied to the length of each of the stages they represent, which for land cover types on the Pevensey Levels are reproduced in Table 4.5. The resulting $Kc_{\text{Crop Type}}$ curve is then varied according to soil moisture characteristics to produce a time series of PET_{Crop}, representing the multiplication factors required for the estimation of AET throughout the year.

Numerous models are available for the adjustment of the $Kc_{\text{Crop Type}}$ curve, representing the second step of the FAO approach. The UK Ministry of Agriculture, Fisheries and Food [MAFF] (1967) suggest a simple model analogous to that incorporated in FAO Crop Water Requirements model (CROPWAT) (Smith, 1992). These models assume that the soil profile contains 125 mm of water at field capacity (*FC*). AET is subsequently calculated based on Penman PET_{Crop} combined with an index of wetness relative to field capacity. For the first 50 mm of available soil moisture PEt_{crop} is assumed equivalent to AET ($Kc_{\text{Water Availability}} = 1$). As the soil dries out AET is reduced relative to PEt_{crop} , so that for the for the next 50 mm $Kc_{\text{Water Availability}} = 0.5$ and for the final 25 mm $Kc_{\text{Water Availability}} = 0.25$. Similarly, Brereton *et al.*, (1996) suggests that $Kc_{\text{Water Availability}} = 1$ when soil moisture deficit (SMD) is less than 40mm. At greater deficits, $Kc_{\text{Water Availability}}$ is given by

$$Kc_{\text{Water Availability}} = 1.5 - 0.0125SMD \qquad \text{(Brereton } et\ al.\text{, 1996)(Equation 4.3)}$$

although in this approach, the Priestley-Taylor method is employed to provide estimates of PET_{Ref}. An equivalent two-step model is incorporated into the Meteorological Office Rainfall and Evaporation calculation System (MORECS) for the estimation of AET. Net radiation, temperature, vapour pressure and windspeed are monitored at meteorological stations across the UK, and using objective interpolation employed to obtain 40x40 km grid square values of PEt_{Crop} by the Penman-Monteith method. AET can then be calculated using a soil moisture extraction model by *'progressively reducing the actual rate of water loss from the potential value to zero, as available moisture decreases from saturation to 0'* (Hough *et al.*, 1997).

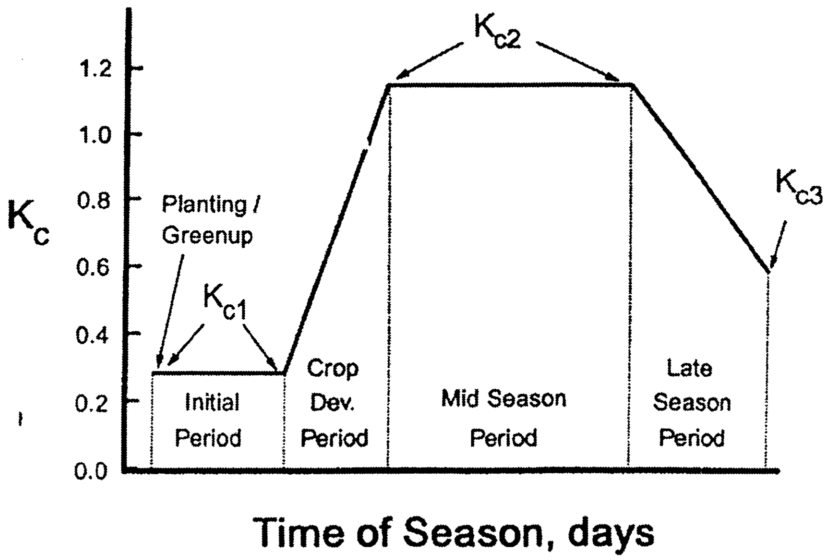


Figure 4.1. Food and Agriculture Organisation (FAO) crop coefficient curve and stage definitions.

Initial	*Planting to 10 % ground cover* (Highly dependant on crop and time of year)
Crop development	*10% cover* to *effective cover* (effective cover = initiation of flowering for many crops)
Mid Season	*Effective cover to start of maturity* (start of maturity is often indicated by leaf yellowing or senescence)
Late season	*Start of maturity* to *harvest*

Table 4.4. General benchmark growth stages for defining FAO crop curves (from ASCE, 1996).

Vegetation Type	Initial	Development	Mid-Season	Late Season
Grass pasture (rotation)	10	20	-	-
Cattails, bulrushes	10	30	80	20
Short Vegetation (0.3m)	180	60	90	30
Corn	25	40	45	30
Winter wheat	20	70	40	25
Barley, wheat and oats	15	30	65	40

Table 4.5. Lengths of crop development stages (in days) for land cover types present on the Pevensey Levels wetland (from Doorenboos and Pruitt, 1977).

4.4. The Penman-Monteith method

An important distinction between the MAFF/FAO and MORECS approaches reviewed in Section 4.3 is the model initially employed to obtain PET_{Ref}. Recently, the Penman-Monteith method (PET_{P-M}) has superseded that the use of Penman within the MORECS approach, giving more consistent PET estimates, and performing better than other reference methods when compared with lysimeter data (Smith, 1992, Chiew *et al.*, 1995). PET_{P-M} is given by

$$PEt_{P-M} = (\Delta\,(R_n - G) + \rho C_p\,(e_s - e) / r_a) / (\Delta + \gamma\,(1 + r_s/ r_a)) \qquad \text{(Equation 4.4)}$$

where

R_n is net radiation (kJ m^{-2} s^{-1}),

G is soil heat flux (kJ m^{-2} s^{-1}),

ρ is atmospheric density (kg m^{-3}),

C_p specific heat of moist air,

e_s-e is vapour pressure deficit (kPa),

Δ is the change of saturated vapour pressure with temperature (kPa °C^{-1}),

γ is the psychrometric constant (kPa °C^{-1}),

r_s is crop canopy resistance (sm^{-1}) and

r_a is aerodynamic resistance (sm^{-1}),

The Penman-Monteith model is an extension of Penman's in that it explicitly includes the aerodynamic resistance, r_a, rather than the simpler wind run of the Penman equation (Loomis and Connor, 1992). In doing so, it accounts for the effects of crop-induced turbulence on evapotranspiration. Turbulence is initiated by non-uniformity at the surface, where the interaction of moving air with a rough surface gives rise to mixing, which is a very effective mechanism for transferring water through the atmosphere away from the surface (Shuttleworth, 1979). By including the surface resistance, r_s, the PET_{P-M} method also accounts for the characteristics and stomatal behaviour of the crop canopy, including their size, distribution and the proportion of each day during which they are open, all of which are important influences on photosynthesis. In this way, r_s can be used to show that biological responses can offset increases in atmospheric evaporative demand and that the evaporation rate can reach a limiting value, or even decline, in spite of increases in available energy (Stewart, 1989).

The greatest advantage of the PET_{P-M} method is that, by assigning roughness and surface resistance values to fit various crop types and heights, its application can be extended to a large variety of surfaces as well as soil moisture conditions (ASCE, 1996). If accurate r_a and r_s estimates are available, the PET_{P-M} method is capable of simulating the effects of vegetation and supply of water on ET, the two most important factors in ET rates over wetlands (Lafleur, 1990), making the need for a two-step crop coefficient model redundant. However, the great variation in the stomatal resistances throughout the canopy, within leaves, between leaves and between canopy layers make measurements unreliable (Cain, 1998).

Nevertheless, numerous values for a variety of crop types have been proposed. For wetland and agricultural land cover types on the Pevensey Levels appropriate data are reviewed in Table 4.6. Because of the fore-mentioned limitations, it is common for an adjusted version of the Penman-Monteith equation to be employed by assuming a fixed canopy resistance of 70 sm^{-1} and a crop height of 0.12 m. PET_{Ref} for grass by the Penman-Monteith method is then given by:

$$0.408\ (\Delta\ (R_n - G) + \ \gamma\ 900/T+273)\ /\ (\Delta + \gamma\ (1 + 0.34U_2)) \qquad \text{(Equation 4.5)}$$

where the terms are equivalent to those in equation 4.4. PET estimates derived using Equation 4.5 can then be used for application within the traditional crop coefficient approach. The assumptions incorporated into equation 4.5 indicate that this model is suitable for the estimation of PET_{Ref} on the Pevensey Levels and other wet grassland areas in the UK. In actively grazed wet grasslands, crop height will be similar to the value of 0.12 m employed in equation 4.5, and r_s estimates provided for grass, pasture, and wet grassland, suggest a value of r_s = 70 sm^{-1} as appropriate for these habitats (Gavin and Agnew, 2000, Lansley, 1998).

Authors	Vegetation	R_S (sm^{-1})	Notes
Grass and Pasture			
[1]Szeicz and Long (1969)	Grass (0.15m)	80-120	
Pruitt (1960)	irrigated grass (0.10–0.12m)	40-60	
Kelliher *et al.* (1993)	Grassland	40-50	Minimum daytime values for LAI 1.3
Stewart and Verma (1992)	Grassland	40	Minimum daytime values for LAI 3
Hough *et al.* (1997)	Grass, riparian land	80, 80, 60, 50, 40, 60, 60, 70, 70, 70, 80, 80 (Jan-Dec)	Data employed in MORECS
Oke (1987)	Open water	0 (r_a = 200)	
	Short grass	70 (r_a = 70)	
[1]Kim and Verma (1991)	Grass	80-330	Well watered
		20–100	Moisture stress conditions.
Szeicz and Long (1969)		April 20, May 110	
		June 130, July 130	
		August 50, September 30	
[1]Jaworski (1991)	Grass	June 90 – 140, July 190 – 530	
		August 190 – 1360,	Values describe a dry year
		September 1810	
[1]Russell (1980)	Grass	April 40, May 30 – 50	Based on data for the years 1970, 1971, 1972 and 1973.
		June 0 – 170, July 20 – 160	For all months maxima were in 1970, a dry year
		August 20 – 100	
[1]Stewart and Gay (1989)	Grass	22nd June 100 (0600), 100 (1800)	After overnight rain
		25th June 130 (0600), 250(1800)	Dry surface
		60, 100, 410	Min., Mean, Max.
[1]Jones (1992)	Grass	110, 180, 320	Min., Mean, Max.

Table 4.6. Typical values of surface resistance (r_s) for vegetation types found in wet grasslands in the UK ([1]from Cain, 1998).

Authors	Vegetation	R_S (sm^{-1})	Notes
Wetland Vegetation			
Hough *et al.* (1997)	Bare soil	100	Data employed in MORECS
	Water	0	
Campbell *et al.* (1997)	Raised *Empodisma* peat bog	150	Partially wet
		608	After prolonged dry period
Lansley (1998)	Wet grassland	24 - 106	Pevensey Levels
Gavin and Agnew (2000)	Wet grassland	8 - 155	North Kent Marshes
Arable Crops			
[1]Hough *et al.* (1997)	Winter Wheat	81, 81, 81, 64, 50, 45, 93, 29, 100, 89, 89, 81 (Jan-Dec)	Data employed in MORECS
[1]Hough *et al.* (1997)	Spring Barley	100, 100, 100, 100, 51, 45, 93, 29, 100, 100, 100, 100 (Jan-Dec)	Data employed in MORECS
[1]Russell (1980)	Barley	April 40 10 80	Data for each month are for 1970, 1971 and 1972 respectively. 1970 was a dry year. Seasonal changes are due to change in LAI.
		May 150 20 30	
		June 150 20 30	
		July 160 90 70	
		August 200 90 80	
[1]Kim *et al.* (1989)	Wheat	20th May 120 80 100	Values are 0800, 1700 respectively. LAI 6.5
		7th June 240 120 170	
[1]Kim *et al.* (1989)	Barley	19th June 80 170 250	Values are 0900, 1300, 1800 respectively. LAI 2.8
		25th June 170 200 250	
		28th June 140 250 250	
		2nd July 140 250 330	

Table 4.6. Continued.

4.5. Limitations of the crop coefficient approach

Although widely applied, the suitability of the crop coefficient method has come under increasing scrutiny in recent times. For agricultural crops there is a pronounced spatial variability of data (Stewart, 1989) and observed coefficients are more erratic than the smooth curves generally suggested (Agnew, 1981). The need for these further levels of empiricism is therefore a clear indication of the limitations of the fundamental approach of beginning with PET_{Ref} and trying to correct it to obtain AET (Wallace, 1991). Furthermore, unless the local environment is taken into account, the estimation of PET_{Crop} can be subject to errors of up to 35% (ASCE, 1996), supporting the development of *K*c values specific to individual locations, or land cover types within the same climatic zone. In the context of wet grasslands, the validity of the method is also questionable. Because the *K*c method has its roots in irrigation scheduling design, fewer detailed $Kc_{\text{Crop Type}}$ data are available for natural vegetation than for crops of commercial importance. Indeed, to the authors knowledge, Table 4.2. schematises available crop coefficients for wetland land cover types, but data describing agricultural crops represent only a fraction of those available (see Doorenboos and Pruitt, 1977).

Mitsch and Gosselink (1993) point to the dangers of applying standard ET models based on meteorological variables to wetlands. This is supported by hydrological studies in wet grasslands where the direct measurement of AET has been undertaken (Herbst and Kappen, 1999; Gavin and Agnew, 2000). Of particular importance is the fact that although existing $Kc_{\text{Water availability}}$ models account for the effects of water stress on AET (Section 4.3), they preclude the effects of soil saturation, a common feature of wetland hydrology. Studies by Priestley and Taylor (1972) and Crundwell (1987) have suggested that where the soil surface is saturated, or open water areas are associated with some aquatic vegetation cover, AET may proceed at a rate greater than that estimated on the basis of PET_{Crop} alone. Advective processes may also be responsible for the excess of AET over PET (Ingram, 1983; Herbst and Kappen, 1999; Schellekens *et al.*, 1999). These findings are in conflict with assumptions made by traditional crop coefficient models, where AET is defined as a rate which is *'equal to or smaller than crop evapotranspiration'* (Doorenboos and Pruitt, 1984).

Values ranging from 1.6 and 1.8 have been reported by the ASCE (1996) for stands of bulrushes and cattails surrounded by grass pasture. Factors as high as 2.5 and 3.1 *for Typha latifolia* and *Potomageton nodosus* respectively reported by Crundwell (1987). However, values that are especially high should be evaluated in the context of the experimental design employed to derive them. For example, measurements from swamp tanks are typically made at the edge of swamps and may provide values greater than from an extensive swamp due to advective effects (Linacre, 1970; Oke, 1987). The fact that these coefficients are in excess of unity are based on the theoretical premise that the rate of ET from a vegetated body of open water will proceed at a rate which combines both direct evaporation from the open water area and transpiration from the vegetation. ET from the vegetation will approximate the maximum reference rate at the given energy input, with open water evaporation superimposed upon this figure.

Further interest in the crop coefficient method is related not only to the quantitative aspects of the relationship between AET and PET_{Crop} , but also with respect to the assumption made regarding the way in which AET and PET_{Crop} are related. The MAFF, CROPWAT and MORECS models all operate on the premise that, for a given set of atmospheric conditions, AET and PEt_{crop} can be related to water availability in the manner shown in Figure 4.2.a. Morton (1983) however, has contradicted this traditional notion, proposing that AET and PEt_{crop} are related in a complementary manner (Figure 4.2.b). In this model, any increase in AET due to water supply is matched by an equal, and complementary, decrease in PET_{Crop} , until AET = PET_{Crop} . This relationship has been proved in various environments, including semi-arid short-grass prairies in southern Alberta, Canada and in the Sanguere area of Cameroon (Morton, 1986). Plots of AET and PET as a function of annual rainfall in the catchments of the Laweya, Tuchila, Lilongwe and Rivi-Rivi rivers in Malawi, as well as for unspecified river basins in Puerto Rico, follow practically the same pattern which is indicated by Morton's model (Kovacs, 1987). This study examines the suitability of traditional AET : PET relationships in a wet grassland wetland, while considers the possibility of there being a complementary relationship.

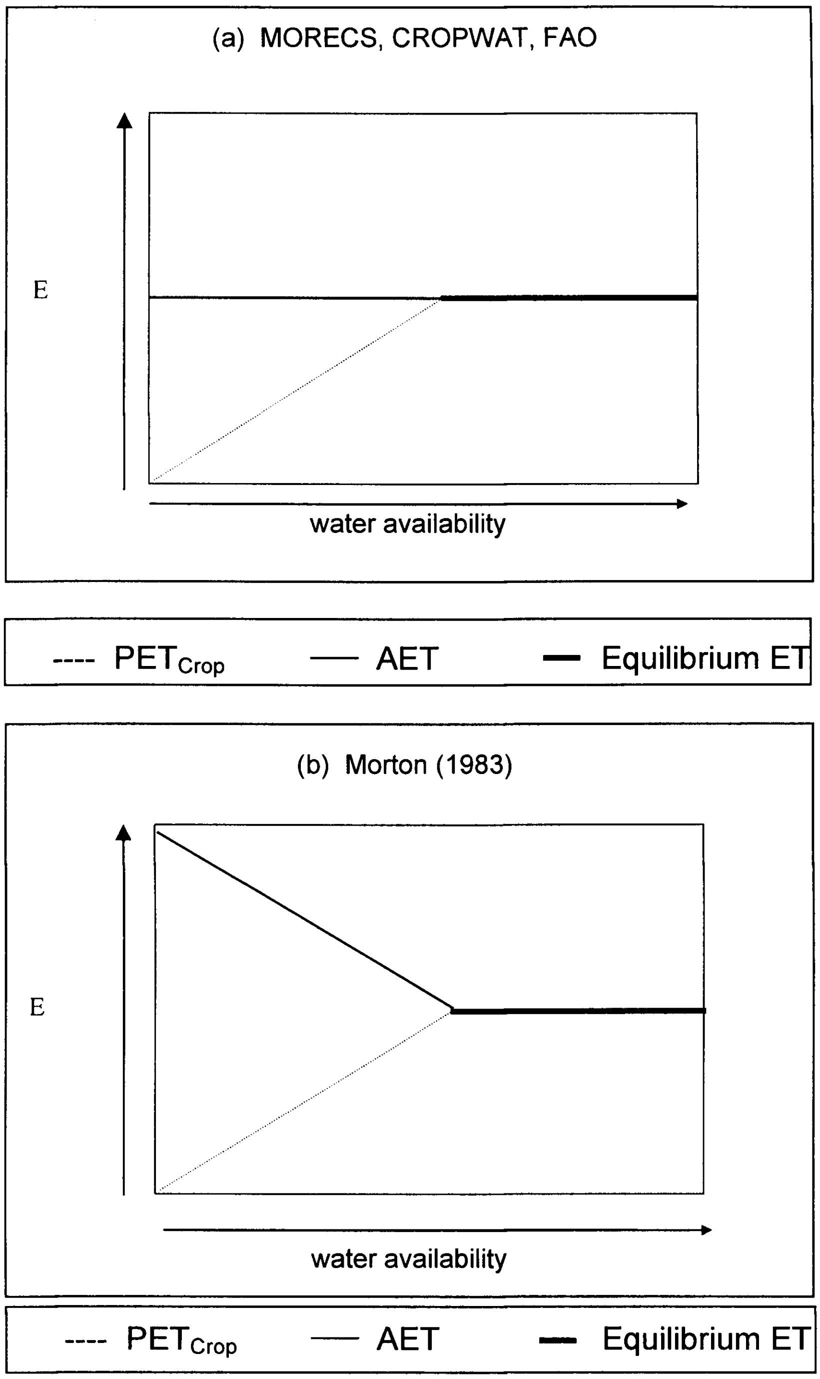


Figure 4.2. Conceptual representation of the two main schools of thought regarding the relationship between AET, PET and water availability.

4.6. Actual Evapotranspiration on the Pevensey Levels

The main objective of this chapter is to assess the validity of the crop coefficient approach in the context of ET estimation in wet grassland environments. The need for reliable estimates of AET from the Pevensey Levels is related to the fact that, in terms of the wetland water balance, the process represents the most significant mechanism of water loss at the wetland scale and may therefore help explain the negative residuals apparent during the summer months (see Section 3.7.4). Analysis conducted in this chapter provides a method for the calculation of AET, based on estimates of PET_{Ref}. In doing so, it represents a single step model for the estimation of evaporative losses from the wetland because values of $Kc_{Crop\ Type}$ are not applied to PET_{Ref} prior to the estimation of AET. The establishment of crop coefficients specific to the Pevensey Levels, are obtained by comparing direct estimates of AET obtained using a state-of-the-art device, PET_{Ref} estimates provided by an Automatic Weather Station (AWS), and hydrological data routinely collected by the Environment Agency (EA). Although not strictly equivalent to crop coefficients shown in Equation 4.2, the coefficients developed are nevertheless termed $Kc_{Water\ Availability}$. This approach is replicated using Horseye PET_{Ref} estimates as a means of evaluating the representativity of ET data traditionally used by the Environment Agency. In later sections, the crop coefficients developed are used to re-evaluate the method employed for the estimation of the ET component of the Pevensey Levels water balance (Section 3.7).

By applying this approach, the importance of accurate evapotranspiration estimation procedures is evaluated in the context of wetland hydrological studies. Results also complement and extend the crop coefficient models currently available for application in wetland environments. In particular, the analysis considers whether the use of traditional PET estimates based on a seasonally invariant coefficient, as employed by both Douglas (1993) and Loat (1994) in previous water balance studies on the Pevensey Levels, provides an accurate approach for the estimation of evaporative losses from the wetland. In doing so, this chapter also furthers the discussion initiated in Section 3.6.1 regarding the suitability of approaches proposed by RSPB *et al.* (1997) for the calculation of wetland water balances. Although these authors identify the need to include ET in water balance calculations, they state that *'it is necessary to assume that the rate of ET for wet grassland is the same as listed for land types listed in MAFF and MORECS bulletins'*. An assessment of the validity of this statement is implicit within the analysis undertaken in this chapter.

4.6.1. METHOD

AET and PET_{Ref} estimates have been examined in association with wetland water availability data, where $Kc_{\text{Water Availability}}$ can be calculated using a one step model by inversion of equation 4.2 and replacing PET_{Crop} for PET_{Ref} by

$$Kc_{\text{Water Availability}} = AET / PET_{Ref} \qquad \text{(Equation 4.6.)}$$

$Kc_{\text{Water Availability}}$ therefore corresponds closely with the notion of relative evaporation (RE) (Gash *et al.*, 1991). Based on traditional models reviewed in Section 4.3, RE should increase with increasing water supply, with the relative magnitude of RE being the $Kc_{\text{Water Availability}}$ coefficient required to calculate AET from PET_{Ref} estimates. Traditional models vary PET based on soil moisture parameters, but the lack of such data on the Pevensey Levels limited the application of this approach. As a result, ditch water levels were employed as a surrogate measure of water availability. In the area where the micro-meteorological instrumentation was sited, ditch water level data were the only measure of daily variations in water availability, the time step chosen for this assessment. These data also represented the longest index of water availability on the wetland, extending back to 1970 for numerous sites and therefore had the greatest potential for future application to historic wetland water balance time series. Consequently, an assessment of the effects of ditch water levels on ET is implicit within the analysis.

Numerous estimates of $Kc_{\text{Water Availability}}$ were calculated. These included values calculated using PET_{Ref} data provided by the AWS, termed AWS PEt_{Ref}, as well as PET_{Ref} inferred from data collected at the local climate station, Horseye. Only the Horseye estimates employed by Douglas (1993) and Loat (1994) in previous water balance studies (Horseye PET_{Penman} and Horseye Eo_{Tank}) were included in the analysis. These were chosen as a means of testing the accuracy of ET estimates historically employed in operational practice. Details of the methods used for the estimation of both Horseye PET_{Penman} and Eo_{Tank} have been provided in Section 3.3.2. For the reasons outlined in Section 4.4, the calculation of AWS PEt_{Ref} was based on the adjusted Penman-Monteith method (Equation 4.4). Input data were provided by the AWS, measuring net radiation, wet and dry temperature, rainfall and wind speed.

An important component of the calculation of $Kc_{\text{Water Availability}}$ was the need to establish the representativity of the Horseye climate station. The station is located on a grassy knowle at 6m OD, whilst the elevation of the marsh surface is generally 2m OD (Blackmore, 1993). More significantly however, there are differences in the way in which input data for the calculation of PET_{Ref} are obtained by the AWS and Horseye. AWS PET_{Ref} estimates rely on the real-time measurement of all the variables required for the estimation of PET_{Ref} on an hourly basis, and scaling up to provide daily estimates. In contrast, Horseye PET_{Ref} estimates are obtained based on temperature, wind run and sunshine hour data collected at 0900 on a daily basis, with net radiation estimated based on sunshine hours using the method proposed by MAFF (1967). It is necessary to note that for AWS PEt_{Ref} estimates soil heat flux has not been considered. Although these are required within the Penman-Monteith method, Soil Heat Flux plates were continually damaged by short-eared voles (a protected species) nesting beneath the logger box and the record was short and discontinuous. However, for most crops, the soil heat flux term is small (circa 1 – 5 % R_n) (Loomis and Connor, 1992; Smith, 1992) although some inaccuracies were expected due to the use of this approach.

4.6.2. THE HYDRA MKII, A DEVICE FOR AET MEASUREMENT

AET was measured using a Hydra mkII (Shuttleworth *et al.*, 1988), an eddy correlation device on loan from CEH Wallingford. The Hydra was sited on the SWT Reserve, in the gravity-drained area of the wetland between June and November, 1996, and between June and October 1997. The instrument sensor head, shown in Plate 4.1, is comprised of a fast-response cup anemometer, an infra-red absorption hygrometer, a fine wire thermocouple and a vertical sonic anemometer mounted on a sensor head at a height of 2.8 m above the ground surface. This is complemented by a REBS net radiometer mounted on the instrument mast 1 m above the ground surface. By correlating vertical windspeed with temperature to give sensible heat flux, humidity to give evaporation flux and horizontal windspeed to give momentum transfer, the eddy correlation method is the most elegant of the meteorological methods, with the minimum of theoretical assumptions and the least dependence on surface conditions (Shuttleworth, 1979).

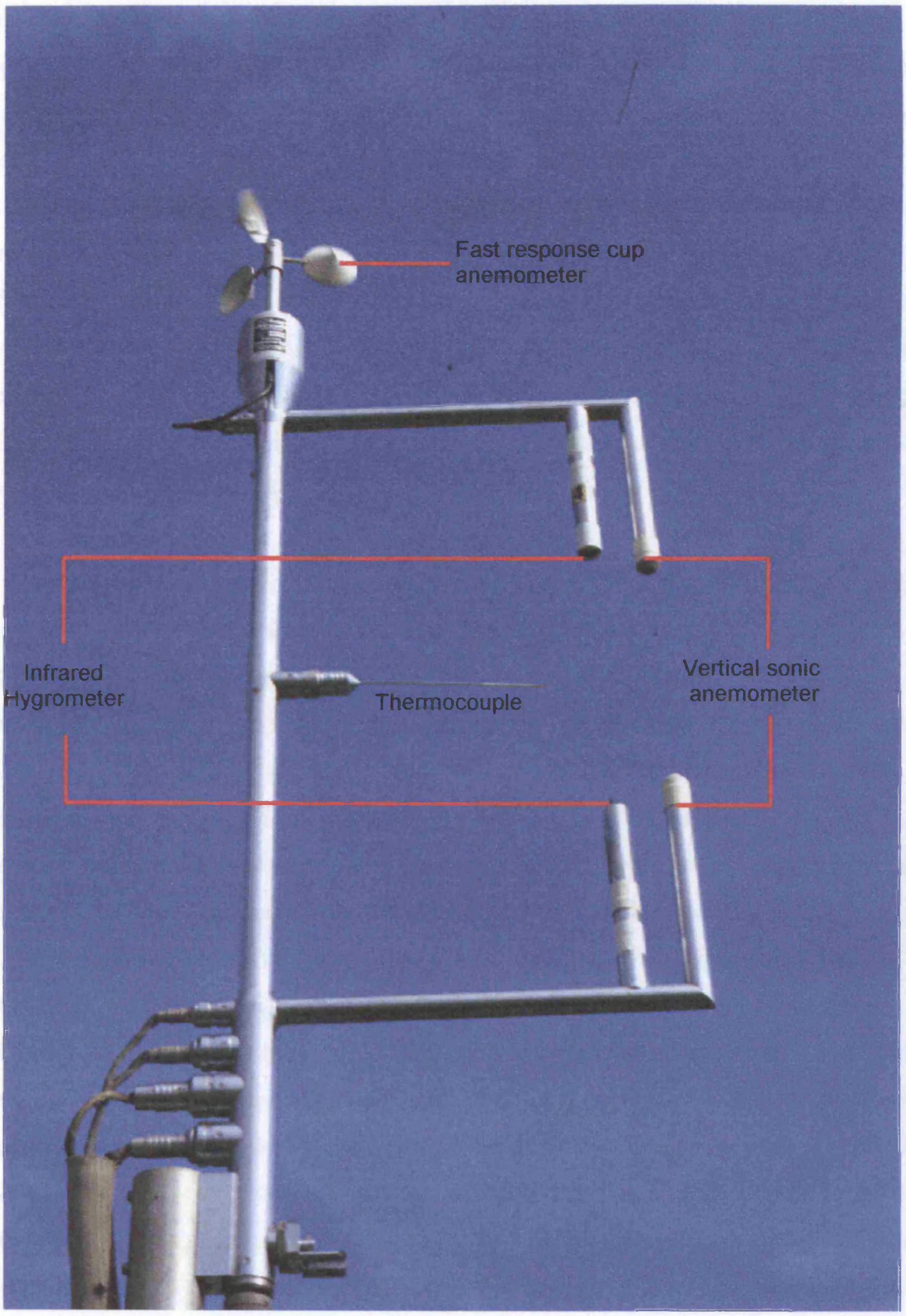


Plate 4.1. Detail of the Hydra mkII sensor head, showing the individual components of the Hydra system.

Like all micrometeorological instrumentation, the Hydra requires an upwind sampling area of uniform undisturbed vegetation or fetch, which for grass of 0.12m height and sensors located at three metres height should be between 200 and 400 metres (Gash, 1986; ASCE, 1996). The flat marsh landscape meant that there was no difficulty in satisfying these requirements. There are however some fundamental technical limitations to the Hydra system. For example it includes sensors of finite size, which generate eddies and turbulence, which are also separated so they may be measuring different eddies (Lansley, 1998). To minimise these problems, the sensor head of the Hydra mkII is designed for minimum aerodynamic interference. All sensors involved in the latent heat flux measurement are positioned within 60 mm of each other to maintain sensor path lengths and provide data regarding the vertical transfer of energy from one eddy to another away from the surface (Shuttleworth *et al.*, 1988).

A second problem is that real-time systems can only compute moving averages which are based on the past behaviour of the measured variable (Shuttleworth *et al.*, 1988). In the Hydra system these problems are addressed by incorporation of computational procedures to correct and calibrate retrospectively (Lansley, 1998). A relative humidity sensor mounted on the instrument mast permits ambient absolute humidity to be calculated in real time for the calibration of the infra-red hygrometer. The performance of the instrumentation can also be checked by comparing the sum of the measured latent and sensible heat fluxes with the available energy (Shuttleworth *et al.*, 1988). This is a particularly advantageous method, as in the Hydra the eddy correlation and net radiation measurements are independent, allowing the accurate validation of data provided by the instrument (Lansley, 1998). Limitations to individual component sensors also require close scrutiny when choosing AET data for analysis. The infrared hygrometer and sonic anemometer do not operate when wet (Shuttleworth *et al.*, 1988). Such errors can be identified in the hourly status value, part of the output data provided by the instrument (Table 4.7). As a result, with experienced installation, operation and quality control, the daily cumulative sum of the evaporation and sensible heat fluxes is normally within 5% of the measured available energy (Shuttleworth *et al.*, 1988).

Month	Date	Year	Time	T	σ T	Rn	σ Rn	W	σ W	U	Sig U	u*	H	E	R	H+E/R	z/L	sW/u*	r_a	Status
September	2	1997	0	9.5	0.25	9.5	0.00	-0.08	0.00	0.4	0.08	-0.02	0	0	-12	0.0	-0.08	0.0	-	0030
September	2	1997	1	9.4	0.23	9.5	0.00	-0.08	0.00	0.7	0.11	-0.05	1	0	-10	-0.1	-0.09	0.0	-	0030
September	2	1997	2	8.7	0.46	9.1	0.00	0.46	0.30	0.9	0.09	-0.12	30	0	-12	-2.5	-1.05	-2.6	-65	0231
September	2	1997	3	8.0	0.16	8.7	0.00	1.04	0.04	0.9	0.10	-0.02	-5	0	-10	0.5	0.19	-1.6	-	0030
September	2	1997	4	8.3	0.38	8.9	0.00	1.03	0.09	1.4	0.09	0.10	-15	0	-9	1.7	0.14	0.9	132	0030
September	2	1997	5	8.5	0.27	9.0	0.00	0.99	0.10	1.7	0.10	0.07	-5	0	-4	1.2	0.02	1.4	341	0030
September	2	1997	6	9.8	0.64	9.8	0.00	0.97	0.10	1.1	0.12	-0.08	6	0	51	0.1	-0.11	-1.2	-	0030
September	2	1997	7	13.4	0.35	12.4	1.15	0.49	0.20	0.6	0.07	0.07	26	154	158	1.1	-3.00	2.9	137	1120
September	2	1997	8	16.2	0.55	13.7	0.76	0.05	0.24	1.4	0.18	0.13	63	182	260	0.9	-0.72	1.8	80	1110
September	2	1997	9	17.7	0.47	11.0	0.81	0.08	0.32	2.7	0.21	0.22	70	252	322	1.0	-0.12	1.4	53	1010
September	2	1997	10	18.6	0.54	10.6	0.78	0.09	0.35	3.6	0.24	0.19	87	268	404	0.9	-0.06	1.8	95	1010
September	2	1997	11	19.2	0.53	10.2	0.74	0.08	0.42	4.5	0.27	0.30	96	305	429	0.9	-0.03	1.4	50	0000
September	2	1997	12	19.1	0.55	11.2	0.69	0.07	0.43	5.1	0.30	0.29	75	272	322	1.1	-0.02	1.5	60	0000
September	2	1997	13	18.9	0.49	10.4	0.69	0.07	0.48	5.5	0.28	0.33	97	299	385	1.0	-0.02	1.4	51	0000
September	2	1997	14	18.7	0.51	11.3	0.64	0.06	0.46	5.8	0.31	0.33	82	246	274	1.2	-0.01	1.4	53	0000
September	2	1997	15	18.2	0.50	12.4	0.52	0.07	0.43	5.6	0.24	0.29	63	193	175	1.5	-0.01	1.5	65	0000
September	2	1997	16	17.8	0.31	12.3	0.40	0.06	0.41	5.3	0.26	0.29	31	124	95	1.6	-0.01	1.4	63	0000
September	2	1997	17	17.0	0.24	12.4	0.64	0.02	0.32	4.5	0.21	0.20	-3	51	-8	-5.8	0.00	1.6	111	1010
September	2	1997	18	15.3	0.49	12.2	1.46	-0.04	0.17	3.1	0.15	0.02	-2	7	-46	-0.1	0.00	8.3	752	2210
September	2	1997	19	13.8	0.25	11.8	2.42	-0.07	0.05	1.9	0.04	0.00	-1	-2	-26	0.1	0.01	999	312	0000

Table 4.7. Sample output from the Hydra mkII system.

4.6.3. STUDY AREA

Both the Hydra and AWS were sited on the National Nature Reserve owned by the Sussex Wildlife Trust, where most of the field-scale hydrological studies described in Section 3.6 have been conducted. The area was chosen for a number of reasons:

- there were no problems with access and the area was sufficiently removed from any roads to be protected from vandalism,
- there was no difficulty in satisfying instrument requirements of between 200-400 metres of undisturbed upwind fetch,
- there were data describing field-scale hydrological conditions since 1995, and
- the area was subject to raised water levels, allowing the assessment of the effects of higher water levels on the magnitude of wetland evapotranspiration loss.

A view of the upwind area from the Hydra is shown in Plate 4.2. Field vegetation is dominated by *Agrostis* spp, although there is considerable *Juncus* spp. in the wetter areas and grips (Plate 4.3.a). The distribution of different land use types in the upwind area, including the distribution of *Juncus* spp. is shown in Figure 4.3. The upwind area was delimited using a contour describing the 400 metre radius around the Hydra, as suggested by Gash (1986), and was characterised by various land cover types, including un-grazed grassland, fields annually mowed for hay, scrapes and ditches (Plate 4.3). Most fields close to the Hydra were actively grazed however, and, as a result, both the Hydra and AWS were placed in a 250 m^2 enclosure to limit damage by stock. Although few data were available, visual evidence suggested that grass length did not vary greatly during the study period, with new growth being rapidly harvested by stock. Ditch vegetation on the nature reserve is extremely rich, consisting of open water, emergent and bank species. The ditches are particularly rich in pondweeds (*Potomageton* spp.) and their surfaces are generally covered during the macrophyte growing season. The ditch directly upwind of the Hydra also showed a profusion of marsh horsetail (*Equisetum palustre*). In terms of water availability, the site is not necessarily characteristic of the wetland as a whole. Ditch water levels are generally higher at this site than elsewhere on the wetland, allowing the evaluation of the effects of wetland restoration strategies such as those reviewed in Section 1.7 on the dynamics of ET.

a)

b)

Plate 4.2. View of the area upwind of the Hydra and AWS.

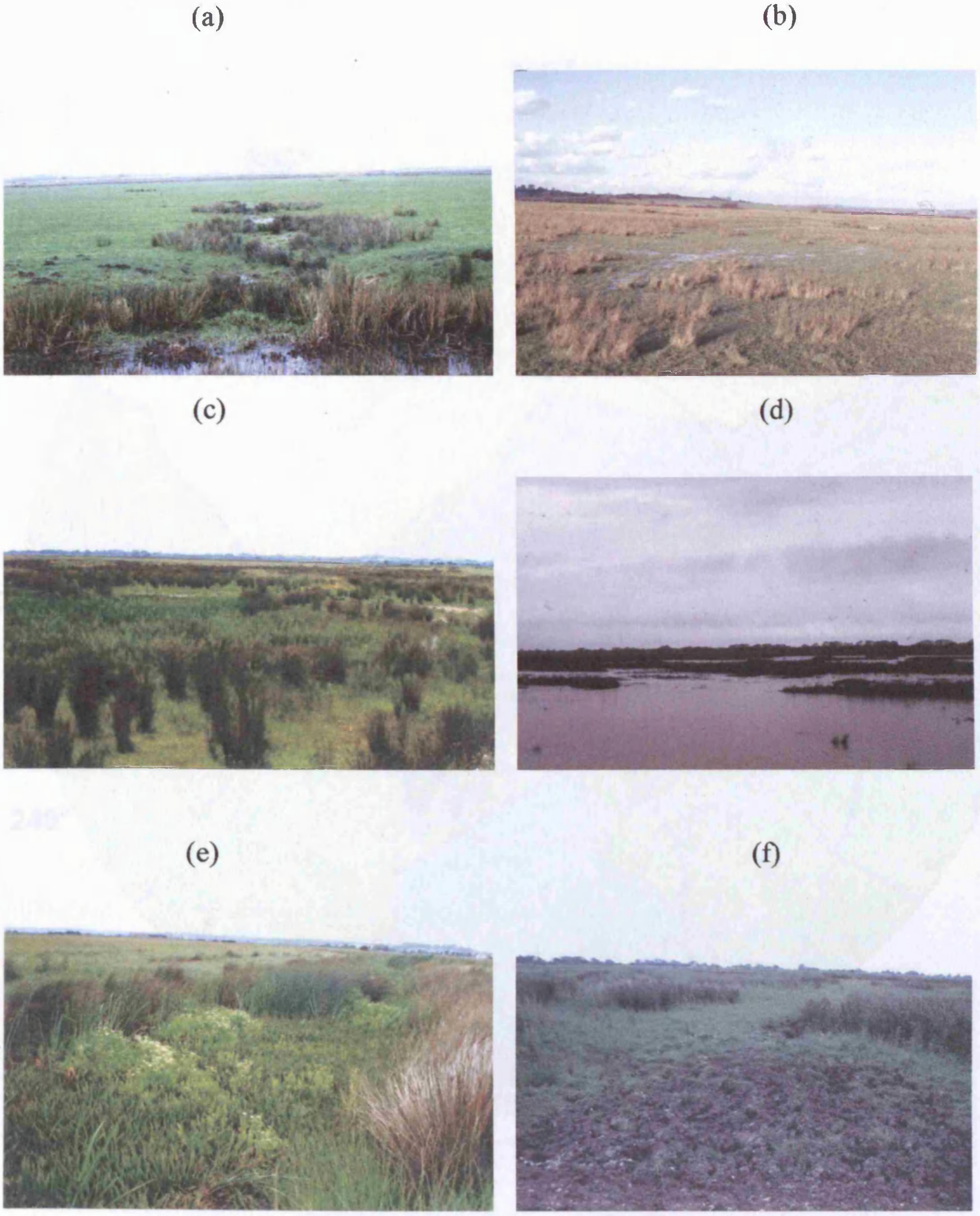


Plate 4.3. Land cover types in the area upwind of the Hydra and AWS vary from (a) dense grass cover (b) intersected by tussocks of *Juncus* in wetter areas such as grips to (c) open water areas (scrapes). (d) In the summer, the scrape is heavily vegetated although it does retain some open water. (e) Similarly, ditches in the summer are almost totally covered by vegetation, (f) although in areas where cattle have trampled the soil, few vegetation grows so that some areas remain as bare soil.

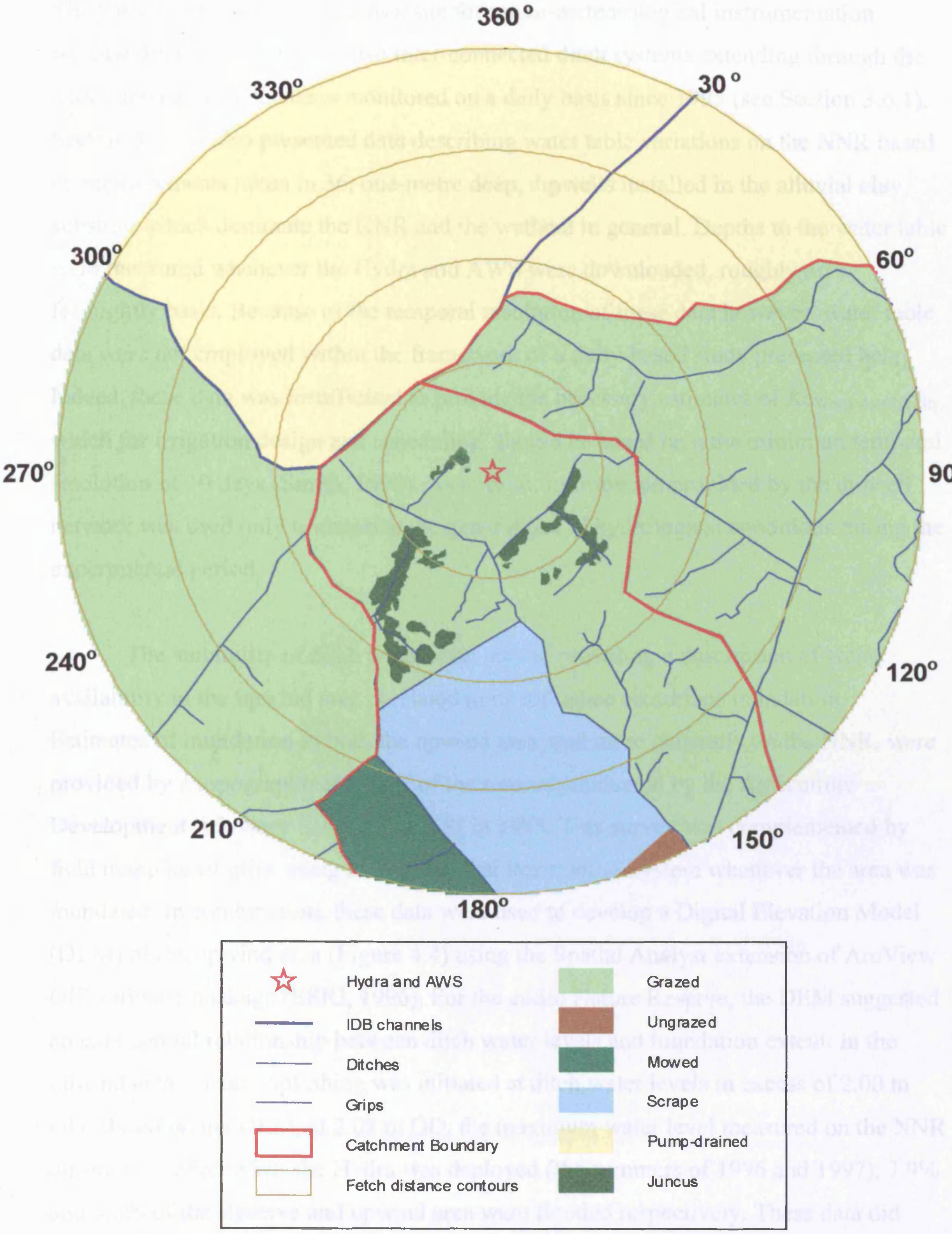


Figure 4.3. Distribution of land cover types and other features likely to influence rates of evapotranspiration rates on the SWT Reserve. Each fetch distance contour shown is equivalent to 100m.

4.6.4. WATER AVAILABILITY PARAMETERS

The SWT Reserve was chosen as a site for micro-meteorological instrumentation because ditch water levels in two inter-connected ditch systems extending through the entire upwind area had been monitored on a daily basis since 1995 (see Section 3.6.1). Section 3.6 has also presented data describing water table variations on the NNR based on measurements taken in 36, one-metre deep, dipwells installed in the alluvial clay substrate which dominate the NNR and the wetland in general. Depths to the water table were measured whenever the Hydra and AWS were downloaded, roughly on a fortnightly basis. Because of the temporal resolution of these data however, water table data were not employed within the framework of a daily-based study presented here. Indeed, these data was insufficient to provide the necessary estimates of $Kc_{\text{Water Availability}}$, which for irrigation design and scheduling, should be based on a the minimum temporal resolution of 10 days (Smith, 1992). As a result, information provided by the dipwell network was used only to describe the generalities of hydrological conditions during the experimental period.

The suitability of ditch water level data in providing a description of water availability in the upwind area is related to its influence on surface inundation. Estimates of inundation in both the upwind area, and more generally on the NNR, were provided by a topographical survey of the reserve conducted by the Agriculture Development Advisory Service [ADAS] in 1993. This survey was complemented by field mapping of grips using a Geographical Positioning System whenever the area was inundated. In combination, these data were used to develop a Digital Elevation Model (DEM) of the upwind area (Figure 4.4) using the Spatial Analyst extension of ArcView GIS software package (ESRI, 1996). For the entire Nature Reserve, the DEM suggested an exponential relationship between ditch water levels and inundation extent. In the upwind area, surface splashing was initiated at ditch water levels in excess of 2.00 m OD. Based on the DEM, at 2.08 m OD, the maximum water level measured on the NNR during the period when the Hydra was deployed (the summers of 1996 and 1997), 7.9% and 6.6% of the Reserve and upwind area were flooded respectively. These data did however assume that all low-lying in-field areas were connected to the ditch system in some way, although the extensive network of grips on field surfaces on the Reserve (Figure 4.4) partially supported this assumption.

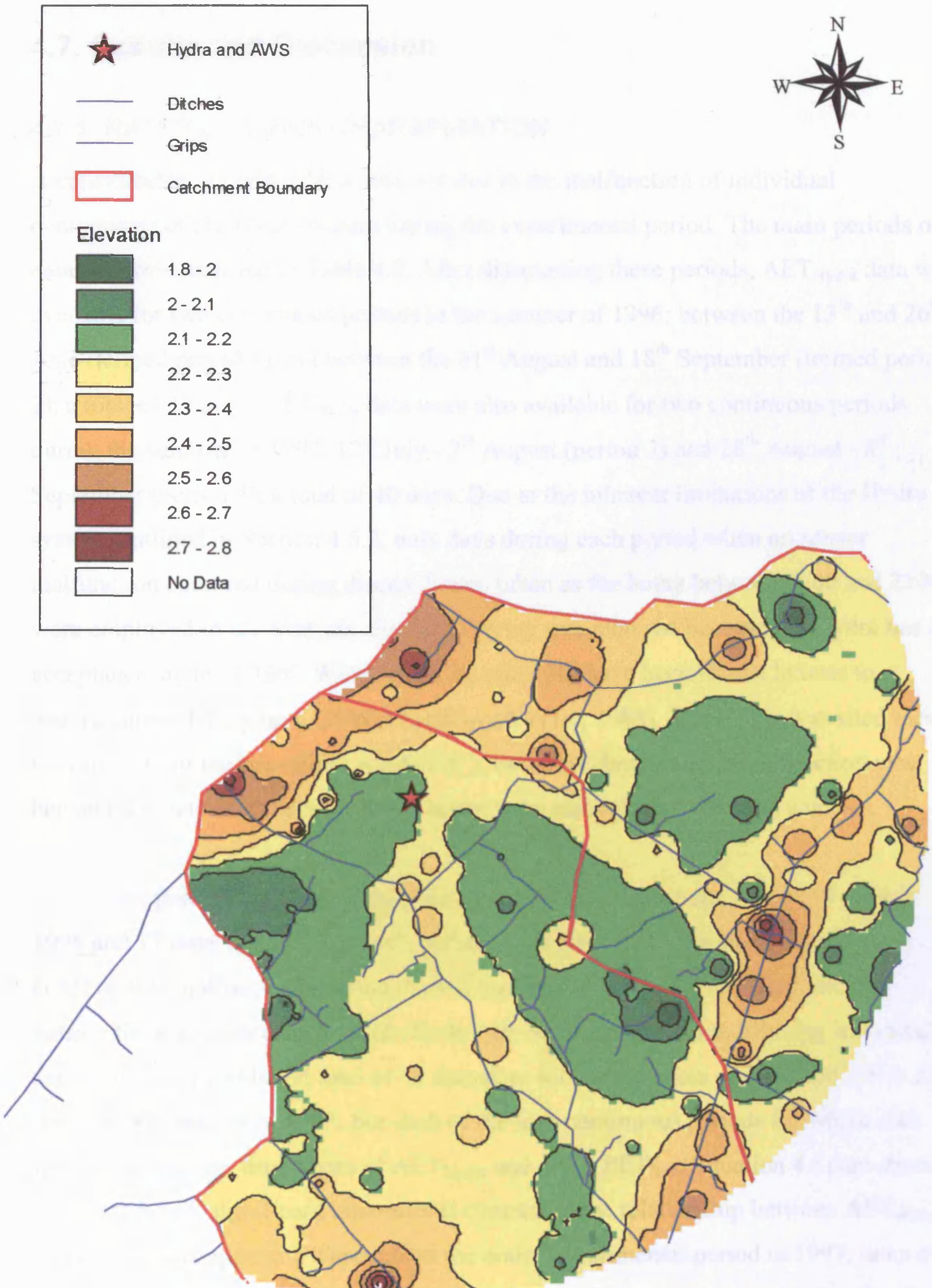


Figure 4.4. Digital Elevation Model (DEM) of the area upwind of the Hydra and AWS (data in mOD).

4.7. Results and Discussion

4.7.1. RATES OF EVAPOTRANSPIRATION

A considerable amount of data was lost due to the malfunction of individual components of the Hydra system during the experimental period. The main periods of data loss are identified in Table 4.8. After discounting these periods, AET_{Hydra} data were available for two continuous periods in the summer of 1996: between the 13th and 26th June (termed period 1) and between the 31st August and 18th September (termed period 2), a total of 33 days. AET_{Hydra} data were also available for two continuous periods during the summer of 1997: 12th July - 7th August (period 3) and 28th August - 8th September (period 4), a total of 40 days. Due to the inherent limitations of the Hydra system, outlined in Section 4.6.2, only days during each period when no sensor malfunction occurred during diurnal hours, taken as the hours between 0700 and 2100 were employed in the analysis. Further filtering was required because the Hydra has an acceptance angle of 330^{o}. Winds from the other 30^{o} have been shown in tests to underestimate ET by up to 10% (Shuttleworth *et al.*, 1988). The Hydra was sited facing the direction of the prevailing wind (190^{o}), therefore days when wind direction was between 25^{o} and 355^{o} during diurnal hours were also rejected from the analysis.

For periods 1 - 4, filtering of AET_{Hydra} data resulted in the loss of 10 days in 1996 and 17 days in 1997. For each period, days discarded from the analysis due to either sensor malfunctions during diurnal hours, or where the wind direction was outside the acceptance angle of the Hydra are shown in Table 4.9. Filtering of available data provided a combined total of 40 data days for the summers of 1996 and 1997: 22 days in 1996 and 18 in 1997. For each of the four continuous periods for which data were available, the time series of AET_{Hydra} and AWS PET_{Ref} (Equation 4.5) are shown in Figure 4.5. A significant inter-annual contrast in the relationship between AET_{Hydra} and PET_{Ref} was apparent. Throughout the entire experimental period in 1997, rates of AET_{Hydra} exceeded AWS Penman-Monteith PET_{Ref}. This contrasted with the data for the summer of 1996, where AET_{Hydra} was generally equal, to or less than, the rate of AWS Penman-Monteith PET_{Ref}. This was illustrated by the mean values of $Kc_{Water\ Availability}$ obtained in 1996 and 1997: 0.96 and 1.67 respectively.

Year	From	To	Sensor malfunction	Parameter affected
1996	1200 28/06	1200 25/07	Thermocouple	Heat Flux (H)
1996	1200 25/07	1300 28/08	Hygrometer	Evaporation (E)
1996	1500 19/09	1100 06/11	Hygrometer	Evaporation (E)
1997	1100 25/06	1000 11/07	Hygrometer	Evaporation (E)
1997	0600 08/08	1400 22/08	Thermocouple	Heat Flux (H)
1997	1300 22/08	1300 28/08	No space in store	ALL
1997	1300 10/09	1300 22/10	Thermocouple	Heat Flux (H)

Table 4.8. Data lost during the summers of 1996 and 1997 due to sensor malfunction.

Year	From	To	Excluded due to hygrometer malfunction	Excluded due to wind direction
1996	13th Jun	26th Jun	20th Jun	-
			21st Jun	-
			22nd Jun	-
			24th Jun	-
			25th Jun	-
1996	31st Aug	18th Sept	-	10th Sept
			-	11th Sept
			-	15th Sept
1997	12th Jul	7th Aug	14th Jul	-
			15th Jul	-
			16th Jul	-
			18th Jul	-
			19th Jul	-
			21st Jul	-
			25th Jul	-
			26th Jul	-
			31st Jul	-
			1st Aug	-
1997	28th Aug	8th Sept	6th Sept	-

Table 4.9. Dates for which Hydra data were available during the summers of 1996 and 1997, and days excluded from the analyses due to either sensor malfunction or wind directions outside the Hydra acceptance angle during diurnal hours.

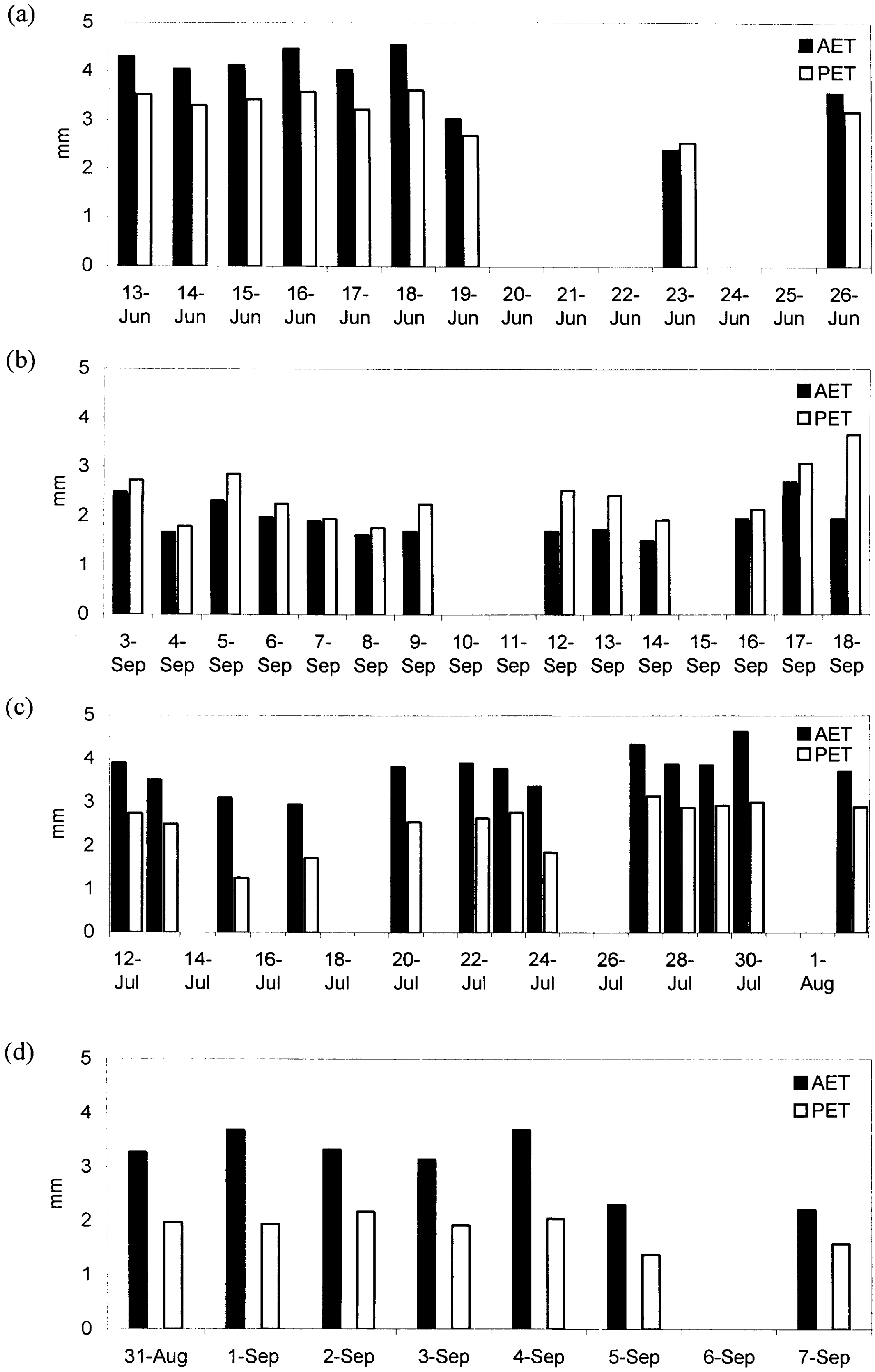


Figure 4.5. Comparison of daily rates of AET_{Hydra} and $PET_{Penman\text{-}Monteith}$ for periods in 1996 (a and b) and 1997 (c and d) for which Hydra data were available.

4.7.2. HYDROLOGICAL CONDITIONS DURING THE STUDY PERIOD

The AET_{Hydra} data available described evapotranspiration during two contrasting summers. The summer of 1996 was considerably drier than average. Rainfall between June and September 1996 was less than half that in the equivalent period of 1997: June to September rainfall in 1997 was 307 mm compared to 147 mm in 1996. June, July and September 1996 were particularly dry, and rainfall was 40%, 48% and 37 % of respective 1961-1990 monthly averages, although August was wetter (Figure 4.6.a). In contrast, June and August 1997 had rainfall 333 % and 203% of respective long term averages (Figure 4.6.b) with 115 mm falling between the 20th and the 27th of June 1997, and 61 mm in the last week of August 1997. In comparison, July and September 1997 were dry, with only 50% and 8% of long term monthly averages respectively (Figure 4.6.b).

The important influence exercised by rainfall on ditch water levels was evident for both summers during the experimental period. Ditch water levels during the summer of 1996, varied from a high of 1.68m OD on the 12^{th} of June falling to a low of 1.25m OD on the 18^{th} September (Figure 4.6.c). In contrast, ditch water levels during the summer of 1997 did not fall below 1.6 m AOD and were above 2.00 m O.D for 27 % of all days during the 1997 experimental period (Figure 4.6.d), reaching a maxima of 2.08m OD on the 7^{th} of July 1997. The higher ditch water levels apparent in 1997 however were not solely a factor of the greater rainfall. Between the summers of 1996 and 1997 a new sluice was installed which allowed the maximum achievable ditch water levels on the SWT Reserve to be raised by 0.4 m (Neil Fletcher, SWT Reserves Manager, Pers. Comm.). Water table data mirrored the inter-annual differences in ditch water levels, and were generally higher in 1997 than in 1996 (Figure 4.6.e and f). At the beginning of June 1996 only dipwells in field 3 contained any water and by July all the dipwells on the reserve were at their dry depth, 0.8-1.0 m beneath the marsh surface (Figure 4.6.e). At the beginning of the summer of 1997, mean water table levels were close to the dry level, but rose to within 0.2 m of the marsh surface at the beginning of July and remained above 0.5 m of the surface for the rest of the summer (Figure 4.6.e). Soil pits dug in the field suggest that, at this level, the water table is within reach of the grass crop, and therefore likely to play an important role in the dynamics of evapotranspiration.

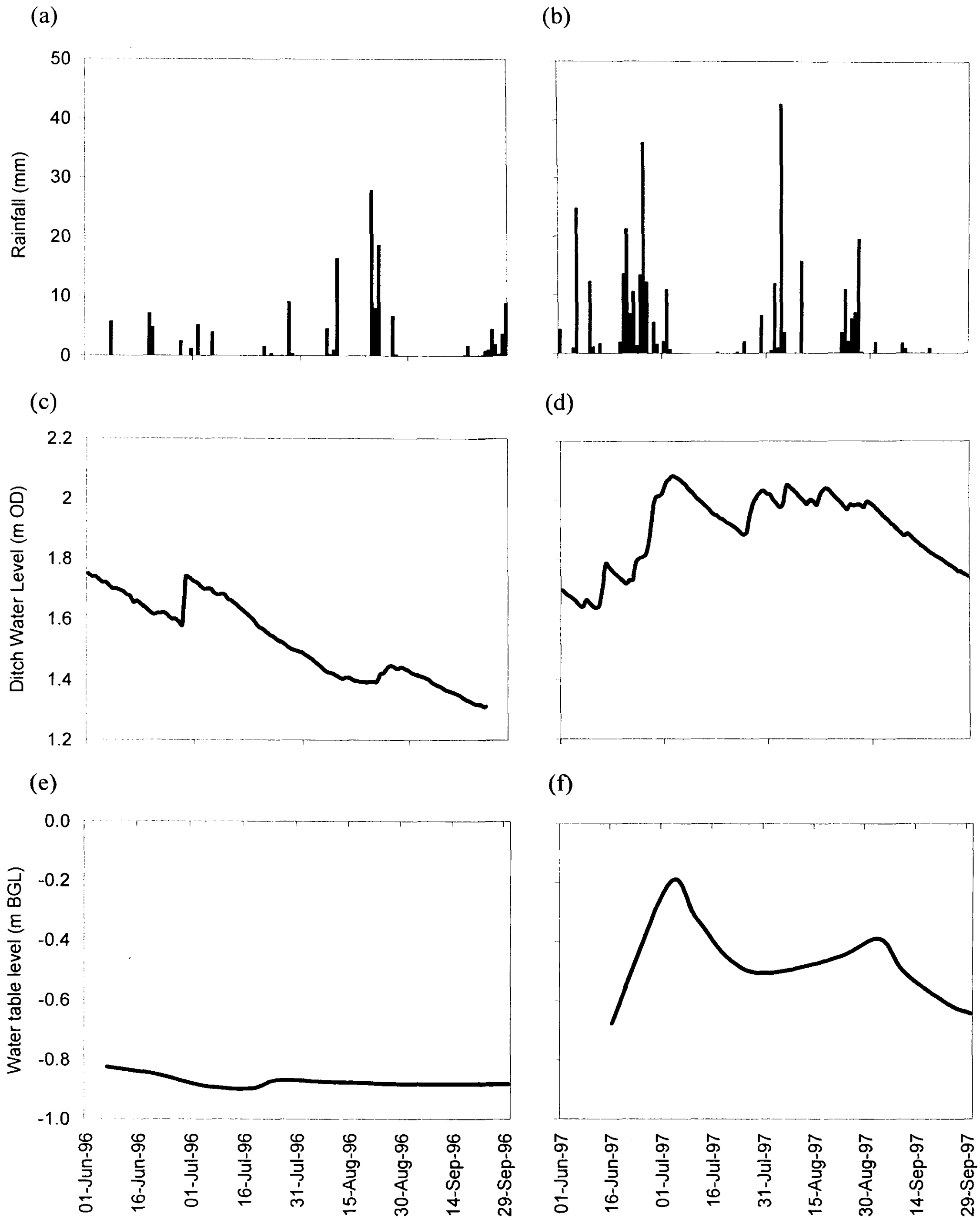


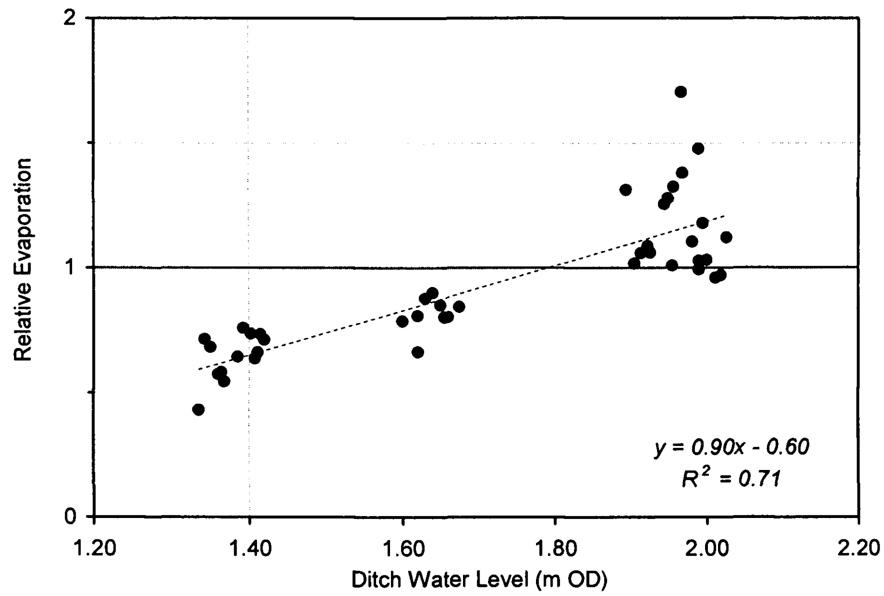
Figure 4.6. Hydrological conditions on the SWT Reserve during the study period. (a) Monthly rainfall at Horseye between June and September 1996 and (b) 1997 relative to the long-term 1961-1990 monthly mean. (c) Daily ditch water levels in the Field 2 ditch system upwind of the Hydra, between June and September 1996 and (d) 1997, and (e) in-field water table levels between June and September 1996 and (f) 1997.

4.7.3. RELATIONSHIP BETWEEN $Kc_{\text{WATER AVAILABILITY}}$ AND WATER LEVELS

The marked differences in AET_{Hydra} and AWS PET_{Ref} time series for 1996 and 1997 provided a preliminary indication of the need to vary crop coefficients applied to wet grasslands based on hydrological parameters. The most obvious difference between the two years were the hydrological conditions in each. Data suggested that the hypothesis that increases in water availability would result in comparative increases in $Kc_{\text{Water Availability}}$, as advocated by traditional crop-coefficient approaches (Section 4.3), was appropriate. Values of $Kc_{\text{Water Availability}}$ for different water availability scenarios were established by correlating daily and five day values, were obtained by application of equation 4.6 and ditch water level data.

Ditch water levels and $Kc_{\text{Water Availability}}$ were related in a linear fashion (Figure 4.7.a and b), as advocated by Brereton *et al.* (1996). The equations describing the relationships are given in Table 4.10. For both daily and five day data, the highest values of $Kc_{\text{Water Availability}}$ were associated with the highest ditch water levels. For daily data, values of $Kc_{\text{Water Availability}}$ above unity were obtained at ditch water levels in excess of 1.78m AOD. For five day periods unity was attained by ditch water levels in excess of 1.76m AOD. Both water levels were in close accordance with the water level at which the DEM suggested the inundation of field surfaces was initiated (*ca.* 1.95m OD; Section 5.2.5). For both sets of data, results indicated that accurate estimates of AET could be provided based on estimates of PET_{Ref} and ditch water levels. The strength of the relationships however was reduced at the highest ditch water levels, particularly when daily data were considered. Nevertheless, the correlation obtained by combining measures of water availability with PET_{Ref} afforded considerably greater accuracy than estimating AET based on AWS PET_{Ref} alone. Although some degree of scatter was apparent in the relationship between $Kc_{\text{Water Availability}}$ and ditch water levels, the correlation coefficients obtained (0.71 for daily data and 0.92 for five day data) were considerably higher than that provided by the relationship between AET and AWS PET_{Ref} (correlation coefficient = 0.21; Figure 4.8). This latter approach is implicit in the approaches proposed for the estimation of ET by RSPB *et al.* (1997), and employed by Douglas (1993) and Loat (1994) in previous water balance assessments of the Pevensey Levels. In both cases, results suggested that more accurate estimates of AET data could be provided by consideration of PET_{Ref} in conjunction with ditch water level data.

(a)



(b)

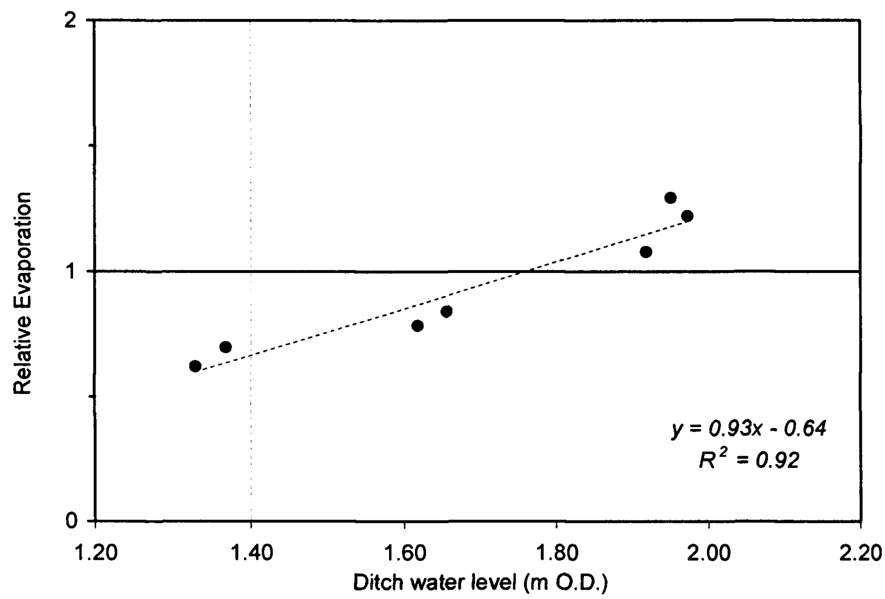


Figure 4.7. The relationship between $Kc_{\text{Water Availability}}$ (shown as the Relative Evaporation) calculated using AWS PET_{Ref}, and ditch water levels for daily and five daily intervals.

PET_{Ref}	Coefficients for the estimation of AET in: *AET = PETRef (DWLa + b)*		R^2	Temporal resolution
	a	b		
AWS $PET_{P\text{-}M}$	0.93	-0.64	0.92	Five day
AWS $PET_{P\text{-}M}$	0.90	-0.60	0.81	Daily

Table 4.10. Coefficients for the estimation of actual evapotranspiration based on AWS PET_{Ref} estimated by the Penman-Monteith method and ditch water levels (DWL) expresses in m OD.

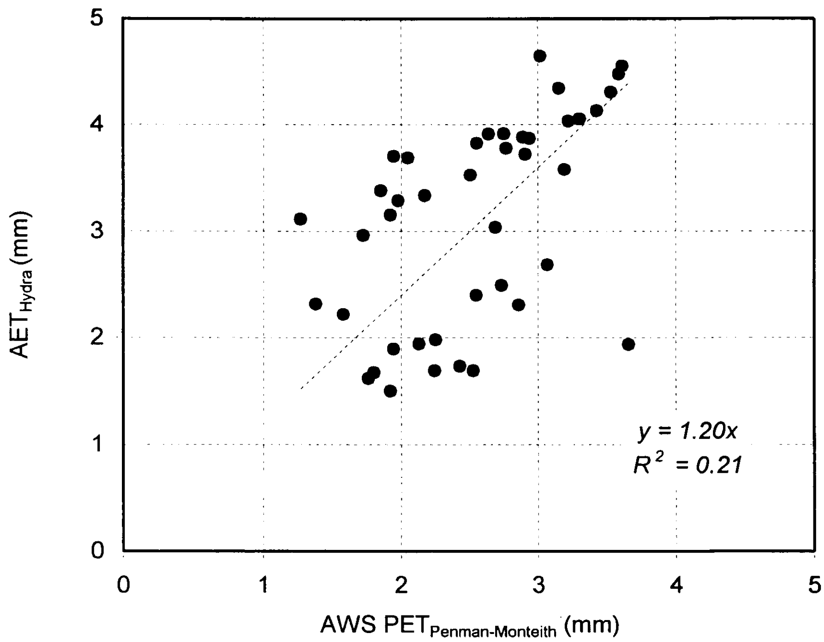


Figure 4.8. The relationship between Hydra AET and AWS PET_{Ref} (both in mm) for all available days during the experimental period.

The scatter evident in the relationship between daily $Kc_{\text{Water Availability}}$ and ditch water levels could be attributed to a number of factors. An important influence was the fact that the characteristics of the sampling area of the Hydra varied from day to day. As wind direction changes, the Hydra will sample ET from a variety of land cover types, which as previously stated vary from grazed, un-grazed, mowed and inundated surfaces (Figure 4.3). A particularly important influence are likely to be the stands of *Juncus* (Plate 4.3) which are in general of greater height than the surrounding grass crop, and are likely to play an important role in determining turbulent exchanges of water vapour between the wetland surface and the atmosphere. The upwind sampling area of the Hydra will also vary in size. Conceptual models of the sampling areas of micro-meteorological instruments provided by Schmidt and Oke (1990), suggest that increasingly stable conditions lead to a shortening of the sampling area, with problems of representativity when comparisons of data from a broad range of atmospheric conditions are considered. Application of the Schmidt and Oke (1990) model to meteorological data obtained from the AWS and Hydra for nine days, suggest that the effective fetch of the Hydra on the SWT Reserve varies between 50 and 400m upwind depending on prevailing atmospheric conditions (Gasca, Hall and Acreman, In Prep.).

Other possible causes for the scatter are related to the adequacy of ditch water levels to simulate hydrological conditions in the upwind area. It is possible that there is not a well defined relationship between ditch water levels and surface inundation, an assumption implicit in the use of ditch water levels to investigate the dynamics of ET. For example, the approach employed here does not account for the effects of inundation other than that induced by ditches, termed 'surface splashing'. Due to the prevailing low hydraulic conductivities of local soils (Section 2.3), surface inundation may also occur by accumulation of rainfall in field surface hollows and depressions. As a result, under some scenarios ditch water level data alone will be insufficient to describe the 'wetness' of the upwind area.

4.7.4. REPRESENTATITIVITY OF HORSEYE ET ESTIMATES

The relationship between daily AWS PEt_{Ref} and Horseye PET_{Ref} data estimated by the tank and Penman methods, shown in Figures 4.9.a and b respectively, suggested that Horseye data over-estimated water loss from the wetland. This supported initial assessments based on monthly ET data, although relationships on a daily basis were characterised by a greater degree of scatter than those presented in Figure 3.8. None of the Horseye PET_{Ref} estimates adequately replicated AET_{Hydra}, as is evident from the scatter apparent in Figures 4.9.c and d. As in the case of the relationship between $Kc_{Water\ Availability}$ estimated using AWS PET_{Ref} and ditch water levels (Figure 4.7), a more accurate means for the estimation of AET was to adjust values based on ditch water levels, especially when tank evaporation data were used (Figure 4.9.f). Nevertheless, the scatter apparent in these relationships was greater than when real-time measurement of meteorological variables were used (*e.g.* AWS PET_{Ref} in Figure 4.7). These differences could be attributed to differences in the methods used for the estimation of Horseye and AWS PET_{Ref}.

The most important difference was the lack of a direct measure of net radiation (R_n) at Horseye. This hypothesis was supported by the relationship between AET_{Hydra} and AWS R_n (Figure 4.10), which identified the dominant role played by R_n on the magnitude of AET. This result is interpreted as an indication of the need to measure R_n directly if accurate AET estimates at Horseye are to be provided. The direct measurement of R_n also provided a potentially more accurate method for the estimation of wetland AET than the application of the crop coefficient approach employing Horseye PET_{Ref} estimates. The correlation coefficients obtained for the relationship between AET_{Hydra} and R_n for the summers of both 1996 and 1997 (0.46 and 0.51 respectively) were generally higher than those provided by the relationships between ditch water levels and both Penman PET_{Ref} $Kc_{Water\ Availability}$ (correlation coefficient = 0.26) and Horseye Tank (correlation coefficient =0.50) (Figures 4.9.e and f respectively). Nevertheless, inter-annual differences in the relationship between AET and R_n between 1996 and 1997 shown in Figure 4.10, suggested that the influence of hydrological conditions was also important. The most likely mechanism was that surface inundation controlled the apportionment of energy to ET due to the need to heat water on inundated field surfaces, although this hypothesis could not be fully evaluated due to the lack of soil heat flux or water temperature data.

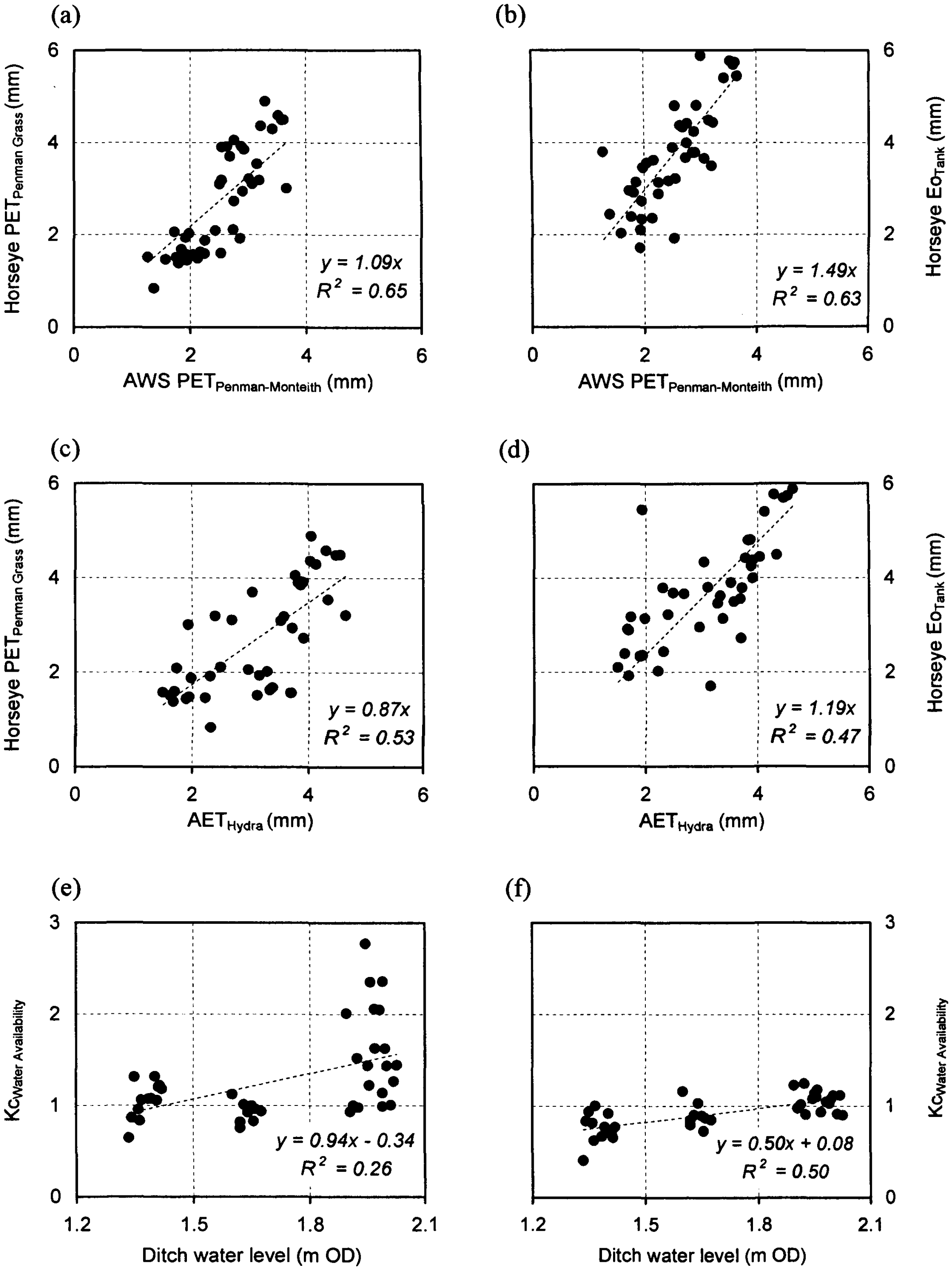


Figure 4.9. The representativity of Horseye PET$_{Ref}$ estimates, evaluated by comparison with (a) and (b) Horseye PET$_{Ref}$ estimates relative to AWS P-M PET$_{Ref}$., (c) and (d) Horseye PET$_{Ref}$ estimates relative to AET. (e) and (f) show $Kc_{\text{Water Availability}}$ models

developed using Horseye $PET_{Penman\ Grass}$ and Horseye Eo_{Tank} estimates respectively (All evapotranspiration estimates in mm).

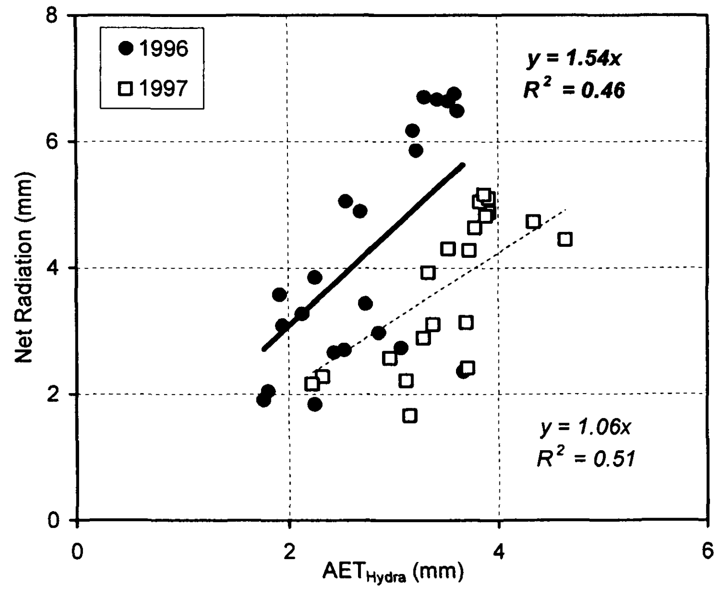


Figure 4.10. The relationship between AET_{Hydra} and net radiation for available days during the summers of 1996 and 1997.

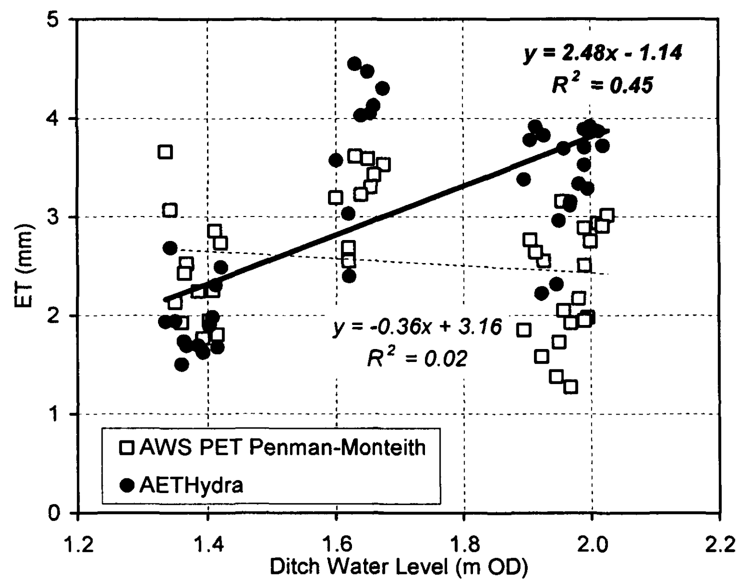


Figure 4.11. Testing Morton's (1983) model: the relationship between AET_{Hydra}, AWS PET_{Ref} and water availability (ditch water levels) on the SWT Reserve.

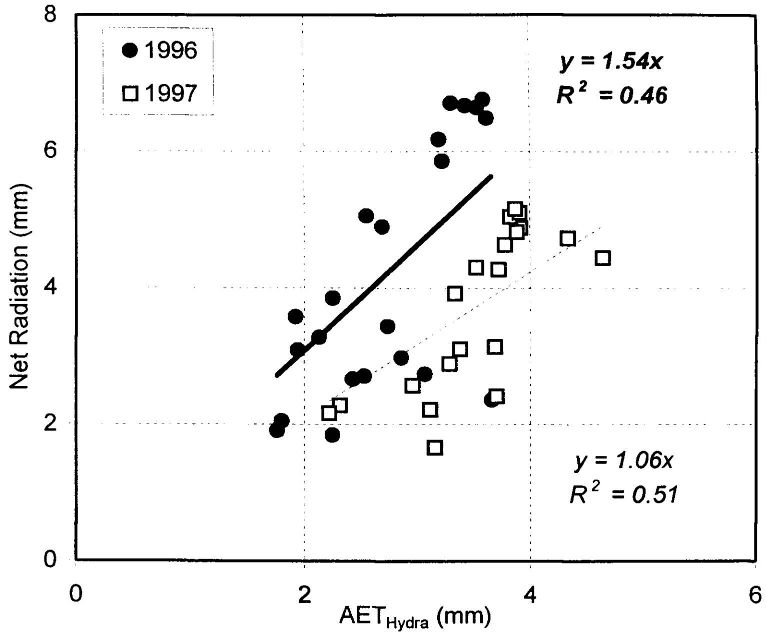


Figure 4.10. The relationship between AET_{Hydra} and net radiation for available days during the summers of 1996 and 1997.

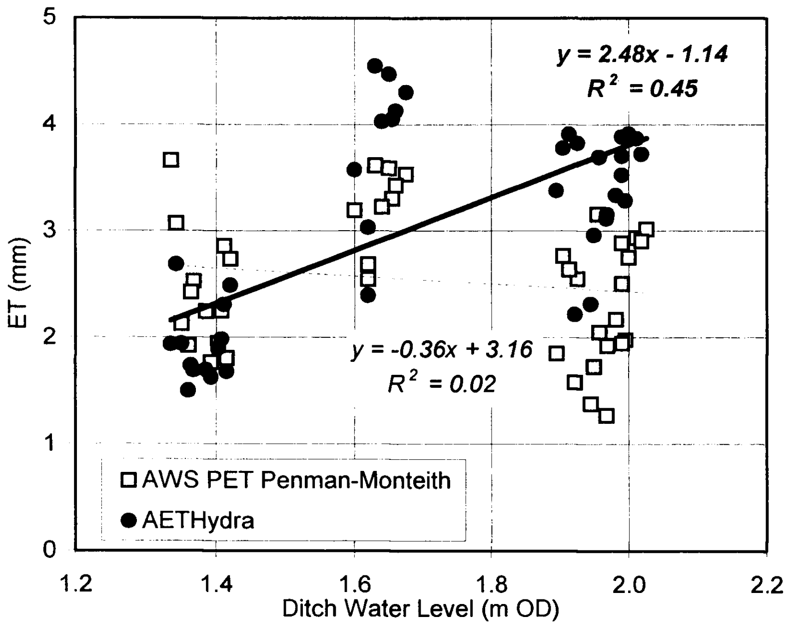


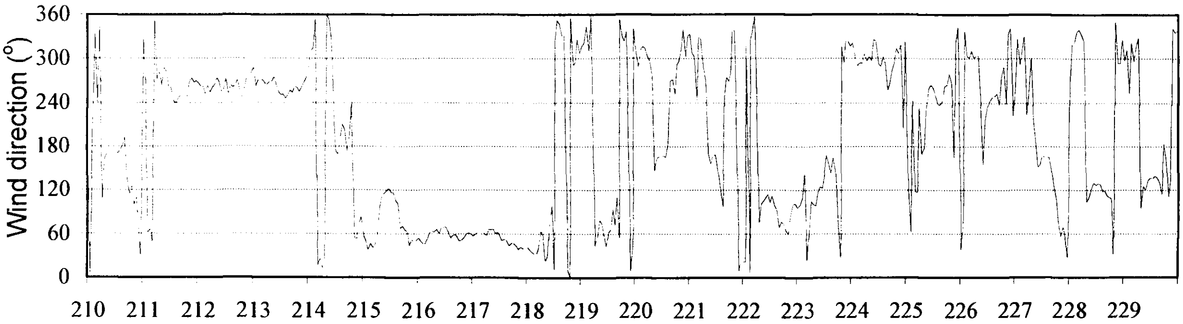
Figure 4.11. Testing Morton's (1983) model: the relationship between AET_{Hydra}, AWS PET_{Ref} and water availability (ditch water levels) on the SWT Reserve.

4.7.5. AN EVALUATION OF THE MORTON AND MAFF APPROACHES

A further issue of interest was the relationship between ditch water levels on the wetland and rates of AET and PET, especially the form of this relationship. Figure 4.2. has illustrated the two main models suggested, the complementary model proposed by Morton (1983) and the traditional model advocated by the MAFF, FAO and MORECS approaches. However, $Kc_{\text{Water Availability}}$ data alone however are incapable of providing an indication of this relationship, since both models, when calculated on a conceptual basis, will result in an increase in $Kc_{\text{Water Availability}}$ with water availability. To test the suitability of the Morton (1983) model, daily rates of AWS Penman-Monteith PET_{Ref} and AEt_{Hydra} were considered relative to ditch water level data. Results are shown in Figure 4.11. An important finding was that the significance of the relationship between AET and ditch water levels was surprisingly high (R^2=0.45), providing further evidence for the influence of hydrological controls on AET. Results had important implications for strategies for the restoration of wet grassland advocating increases in water levels (see Section 1.7.3). Based on the available data, raising water levels on the Pevensey Levels will lead to greater losses by evaporation and evapotranspiration, a factor that should be incorporated in water balance calculations for restored wet grassland sites.

In contrast, the relationship between PET_{Ref} and ditch water level was poorly defined (R^2=0.02). Overall, the analysis suggested that AET and PET were not related in the complementary manner suggested by Morton (1983). Although the relationship between PET_{Ref} and ditch water level was not significant, the form of the relationship was more closely related to Figure 4.2.a than 4.2.b. This supports suggestions by Cain (1998) relating to the inadequacy of the Morton model in areas where extensive advection is a feature of local meteorological conditions. In the prevailing wind direction (270^{o}), the sea is 4 km away and ambient humidity is likely to be influenced by the proximity of the sea. Indeed, the potentially important marine influence is identified in hourly variations in wind direction apparent in some of the daily AWS records, indicating the characteristic switching between inshore and offshore winds over 24-hour cycles (Figure 4.12). This feature of local meteorological conditions may also explain some of the scatter evident in Figure 4.7.a and highlight the potential difficulties posed by applying the crop coefficient approach presented in this chapter to wet grasslands in the UK other than those in coastal locations.

(a)



(b)

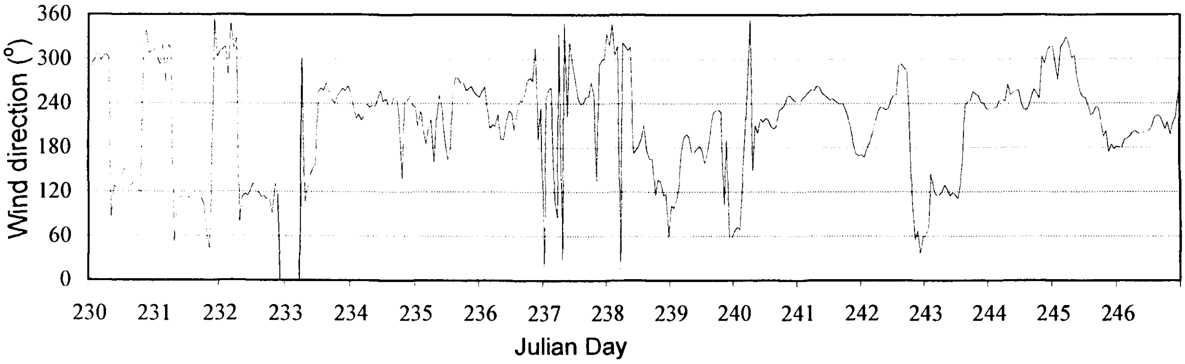


Figure 4.12. Hourly wind direction for periods when data were available during the summers of (a) 1996 and (b) 1997.

PET_{Ref} estimate	Coefficients for the estimation of AET in: $AET = PETRef\,(DWL_{BMFL}\, a + b)$		R^2	Temporal Resolution
	a	b		
Horseye Eo_{Tank}	-0.44	0.99	0.50	Daily
Horseye Eo_{Tank}	-0.41	0.98	0.93	Five Day
Horseye PET_{Penman}	-0.94	1.63	0.26	Daily
Horseye PET_{Penman}	-0.41	1.30	0.31	Five Day

Table 4.11. Coefficients for the estimation of AET from Horseye PET estimates and ditch water level data as a function of the mean field level.

4.8. Re-interpreting the wetland water balance

The findings presented in this chapter are likely to be significant in the context of the wetland water balance, especially relative to the accuracy of water balance calculations, expressed as the water balance residual (Section 3.7.4). The influence of applying different ET estimates within water balance calculations on the accuracy of water balance calculations was evaluated by comparing water balance residuals obtained by the 'traditional' (use of Horseye Eo_{Tank} data coupled with a time invariant coefficient of 0.88) and crop coefficient (Section 4.7.3) methods. These data allowed a more spatially distributed approach to the estimation of losses by ET than had been previously possible because water levels at individual pumping stations could be used to estimate AET from each pumped sub-catchment. The equations presented in Section 4.7.3 for the calculation of $Kc_{Water\ Availability}$ were first computed relative to water levels expressed in m below mean field level. As previously stated, target ditch water levels associated with wet grassland restoration strategies are commonly expressed in this way. An added advantage of this approach was that the adjustment provided a means of standardising the crop coefficient model for application to areas other than the Pevensey Levels.

Another modification when applying the crop coefficient model to the water balance of the Pevensey Levels was that Horseye PET_{Ref} estimates based on evaporation tank data were used to compute $Kc_{Water\ Availability}$. This was because, as stated in Section 3.3.2.3, Horseye tank estimates were the only measure of ET available for the entire water balance period. The standardised equations for the estimation of AET on the Pevensey Levels from daily and five-day Horseye Eo_{Tank} estimates and ditch water levels are shown in Table 4.11. The ET term of the water balance was adjusted based on the application of a five day model applied to mean monthly ditch water level data collected at each individual pumping station (Figure 3.22). As in previous water balance calculations, water level data at the pumping station were assumed to be representative of the entire pumped sub-catchment. The use of a five-day model applied to monthly wetland ET data could be justified due to the limited difference between the slope and intercept associated with the regression between daily and five-day Horseye Eo_{Tank} $Kc_{Water\ Availability}$ models and ditch water levels. These similarities suggest that the crop coefficients developed are applicable through a range of temporal scales.

The general trends apparent in the water balance residual, or error term, have been previously discussed in Section 3.7.4. These can be broadly summarised as negative residuals in 1995 and 1996, especially during the summer months, and positive residuals in 1997 and 1998. The importance of accurate evapotranspiration estimates for wetland water balance studies was clearly apparent from the analysis of Pevensey Levels data. The adjustment of the ET term of the water balance based on the five-day Horseye Eo_{Tank} $Kc_{\text{Water Availability}}$ model and ditch water levels resulted in a considerable reduction in the negative residual of the water balance (Figure 4.13). Over the whole four year period, for months with negative water balance residuals, the adjustment of ET estimates resulted in a mean reduction of 92% in the residual relative to traditional ET estimation approaches. There was however considerable inter-annual variability. Mean monthly reductions of 89% and 83% could be reported for 1995 and 1996, whilst in 1997 and 1998 negative residuals were removed altogether. Adjusted results obtained for the summers of 1997 and 1998 supported the perception that these summers were wetter than average with no water resource deficits. In contrast, for the summers of both 1995 and 1996, although negative residuals were reduced by the $Kc_{\text{Water Availability}}$ approach, the balance between inputs and outputs remained negative.

In general, adjusted ET data suggested that losses from the wetland were consistently over-estimated throughout the entire water balance period when the traditional approach employing the evaporation tank and a time invariant coefficient of 0.88 was used. This was confirmed by comparisons of monthly estimates of wetland ET calculated the traditional and crop coefficient models (Figure 4.14). Conclusions were further supported by data describing annual losses from the wetland by ET using the two methods (Table 4.12). Results could be attributed to the heterogeneous nature of water levels wetland wide. On the Pevensey Levels, water levels in individual sub-catchments operate to create differences in the ET regime across the wetland. A particular difference was apparent between the monthly rates of ET per unit area in pump-drained areas of the wetland and on the SWT Reserve. Due to the higher water levels retained on the SWT Reserve, ET losses per unit area were higher than in pump-drained areas. However, in pump-drained areas, application of the traditional coefficient of 0.88 to tank evaporation estimates resulted in over-estimation of actual evaporative losses (Figure 4.15).

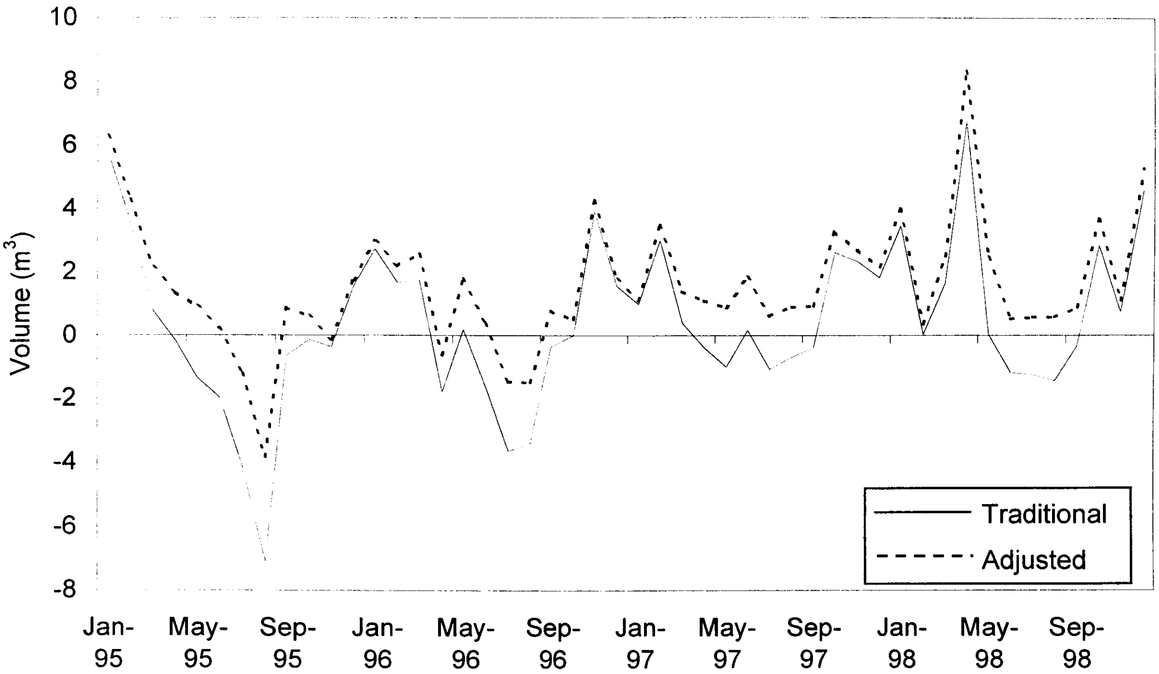


Figure 4.13. Comparison between monthly water balance residuals 1995-1998 obtained by application of a time-invariant coefficient to calculate ET ($0.88PET_{Ref}$, the 'traditional' method) and application of the $Kc_{Water\ Availability}$ model using 5-day Horseye Eo_{Tank} given in Table 4.11 and sub-catchment ditch water levels ('adjusted' method).

YEAR	ET losses by volume (millions m^3)	
	Traditional method	Adjusted method
1995	39.6	22.1
1996	34.0	21.3
1997	35.8	23.1
1998	34.7	20.6

Table 4.12. A comparison between annual ET losses from the Pevensey Levels 1995-1998 estimated by application of a time-invariant coefficient to Horseye Eo_{Tank} data (the 'traditional' method) and application of a $Kc_{Water\ Availability}$ model using Horseye Eo_{Tank} and sub-catchment ditch water levels ('adjusted' method).

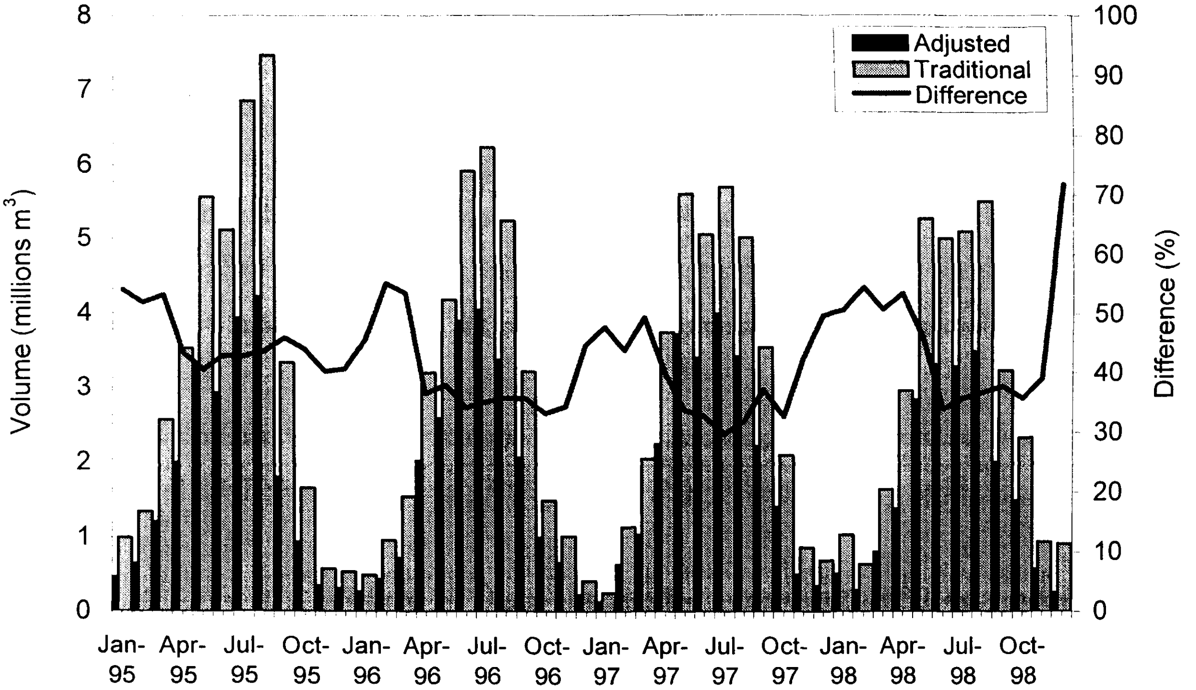


Figure 4.14. Monthly ET losses from the Pevensey Levels 1995-1998 calculated using the 'traditional' approach (0.88 Horseye Eo_{Tank}) and a spatially distributed approach where ET is adjusted according to ditch water levels in each sub-catchment.

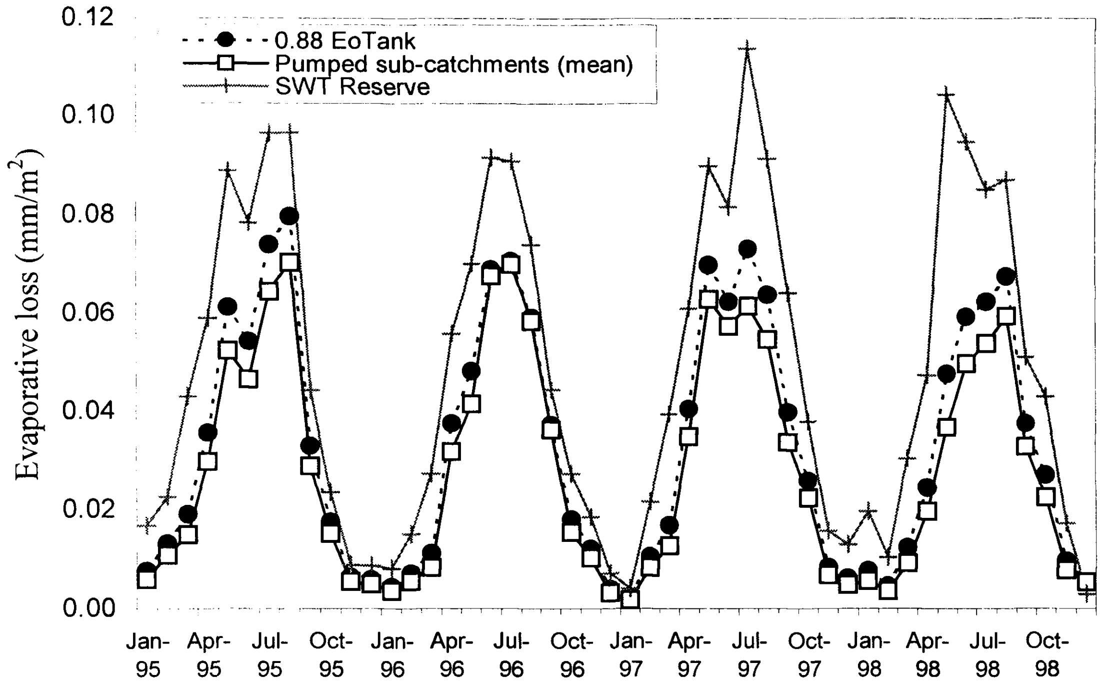


Figure 4.15. Comparison between the monthly rates of ET per unit area (m^2) calculated for pump-drained catchments and the SWT Reserve based on the crop coefficient approach and that estimates as a function of 0.88 Horseye Eo_{Tank}.

The results presented should however be considered within the limitations of the method. Assumptions regarding water levels at the pumping station as representative of the entire pumped catchment may lead to some inaccuracies in the calculation of ET. Limitations to the method also arise because monthly water level data will mask daily variations, which have an important influence on the dynamics of wetland ET. The method also relies on accurate estimates of the mean catchment field level and the data presented by Blackmore (1993) for pumped catchments on the Pevensey Levels wetland are unverified. In any case, some inaccuracies in the calculation can be expected due to the scatter evident in the relationship between $Kc_{\text{Water Availability}}$ (calculated from Horseye Eo_{Tank}) and ditch water levels employed in the analysis and shown in Figure 4.9.f.

Nevertheless, adjusted ET data illustrate the importance of ET as a process at the wetland scale, as well as the large influence the choice of ET estimate can have on calculations of wetland water resource availability. Overall, the results support the need for the detailed measurement of evapotranspiration, as well as the factors most commonly affecting the rates of ET in wetlands. Indeed, the relationship between $Kc_{\text{Water Availability}}$ and ditch water levels, irrespective of the method employed to determine PET_{Ref}, clearly supports the importance of spatially-intensive hydro-meteorological monitoring in wetland environments, especially where the processes controlling ET are heterogeneous in nature.

4.9. Management implications

Results presented in this chapter highlight the importance of accurate evapotranspiration estimation techniques in wetland water balance calculations. On the Pevensey Levels, AET could be inferred from PET_{Ref} estimates provided by the Penman-Monteith method, and ditch water level data as a surrogate of water availability. This method was especially appropriate when climatic data were intensively gathered in real time. Horseye estimates could also be used, although with a smaller degree of confidence in the estimates of AET obtained. The adjusted crop coefficient approach afforded significantly greater accuracy than the use of PET_{Ref} estimates alone to obtain estimates of AET. This latter approach is implicit in traditional methods employed for the estimation of ET loss in water balance calculations previously conducted on the wetland. In these previous studies, time-invariant coefficients have been applied to evaporation tanks or Penman PET without consideration of the influence of water supply on evaporative rates. However, results obtained do support suggestions by Agnew (1981) and Wallace (1991) regarding the limitations of the crop coefficient approach to estimate AET from PET_{Ref} estimates, especially given the degree of dispersion in the daily and five-day relationships between $Kc_{Water\ Availability}$ and ditch water levels for both Horseye and AWS PET_{Ref} estimates.

For the Pevensey Levels, the relationship between AET, PET and water availability, were not related in the form suggested by Morton (1983) (see Figure 4.1b). Trends indicated that the form of the relationship between PET and AET was analogous to that employed by the MAFF (1967), CROPWAT and MORECS models (see Figure 4.1.a). This finding has important implications for ET estimation in UK wetlands. Traditional crop coefficient models, such as those presented in Section 4.3, including those employed by the Environment Agency on the Pevensey Levels and other wet grassland areas, assume a non-complementary relationship between AET and PET. This model has been found to be appropriate in the local context. However, traditional models define AET as the rate of ET '*equal to or smaller than* PET_{Ref}'. Data presented in this chapter contradict this notion, a result that may potentially be applicable in wetland areas other than on the Pevensey Levels.

Values of $Kc_{Water\ Availability}$ at high ditch water levels were in excess of unity when both daily and five-day data were employed. Of particular significance was that for five-day data, unity was attained at a ditch water level approximately equivalent with bankfull conditions in ditches on the study site, and the level at which ditch-induced flooding of field surfaces is initiated. These findings have important implications for wetland restoration programmers where water level manipulation is a feature of revised management strategies. Where PET_{Ref} estimates alone are used to calculate evaporative loss, ET will be underestimated when water levels are maintained at those favoured by birds and enshrined in water level prescriptions associated with, for example the ESA scheme. For example, where ditch water levels provide surface inundation to a depth of 0.2m, advocated by RSPB *et al.* (1997) as ideal for wet grassland waders (see Section 1.7.2.3), results obtained suggest that ET may be underestimated by as much as 49%. In contrast, during dry periods the method will over-represent evaporative losses, supporting a model where ET is constrained by water availability in spite of increases in available energy (see Section 4.4).

This factor should be considered when assessing the sustainability of raising ditch water levels in water resource terms. Results obtained indicate that higher ditch water levels equate to a higher degree of ET loss from the ditch system and, irrespective of the method used, ET represents the main output from the local hydrological balance (Section 3.6.1). The higher rates of ET may also have a bearing on the ability of wetlad manager to satisfy hydrologica,l and therefore ecologica,l targets. This is considered in later sections of this thesis. On the Pevensey Levels, the higher rates of ET observed at high ditch water levels should also be considered when establishing the timing and duration of feeding ditch systems to ensure that target water levels set by restoration schemes can be attained throughout the summer months.

Overall, results presented in this chapter have highlighted the value of combining field-scale and catchment scale hydrological monitoring to reduce hydrological uncertainties associated with changes in the *status quo* of wetland management. The water balance model presented in Chapter 3 and furthered in this chapter is complemented by an equivalent model in the next chapter, which focuses on field-scale hydrological dynamics. As Chapter 3 has collated all data describing catchment scale hydrology, the model presented in Chapter 5 synthesizes all available field-scale data. Both models include the method of ET estimation presented in this chapter, with a view towards examining water resource and hydro-ecological issues at contrasting spatial scales.

CHAPTER 5

MODELLING DITCH WATER LEVELS ON THE PEVENSEY LEVELS WETLAND

5.1. Introduction

Few tools are currently available which allow the wetland manager to assess the potential for, or impacts of, revised water level management strategies in wetland areas. This is particularly the case with respect to the balance between the relative benefits for characteristic wetland biota, and the impacts on farming communities in the areas where they are applied. An issue of equal importance is the generally limited knowledge regarding the availability of water for wetland management schemes, an issue which Chapter 3 has highlighted as a potential limiting factor to their success. In combination, these issues complicate the development of integrated and sustainable wetland management systems in the UK. As 'cultural landscapes' (Section 1.1), these issues obtain increased importance in wet grasslands. Due to the long history of anthropogenic intervention in the natural environment, the maintenance of environmental conditions to support both nature conservation and agriculture is a pre-requisite to retain the social and physical qualities that define wet grasslands as a habitat.

A particular limitation for achieving the 'wise-use' of wet grassland areas has been the limited incorporation of hydrological research within wetland management strategies. This has precluded the use of modelling tools that may be employed to address some of the crucial issues associated with the management of these sites. In particular, the nature of wet grassland environments and the issues associated with their management (Chapter 1), identify hydrology as the crucial basis of any modelling study. The need for hydrological research in the context of wet grassland restoration has been clearly illustrated in previous scientific work conducted in these habitats, much of which has been reviewed in Chapter 1. In the context of the Pevensey Levels wetland, Chapter 3 has provided a preliminary assessment of the effects of raising ditch water levels on in-field water tables, and has considered these relationships in the context of overall water availability, established using the wetland water balance approach. These studies have highlighted the importance of placing field-based wetland restoration within a catchment-wide framework.

This chapter complements the water balance assessment of the Pevensey Levels wetland presented in Chapter 3. The main difference is the treatment of the wetland water balance at the field, as opposed to the catchment, scale. The chapter describes the construction and implementation of a hydrological model designed to simulate ditch water level variations in gravity-drained ditch sub-catchments of the Pevensey Levels. The choice of scale for model development is related to issues raised during meetings of the Pevensey Levels Study Group (Section 2.8.3). These discussions indicate that it is at the field scale where the greatest impacts of revised water level management strategies are apparent. On the Pevensey Levels, the instatement of revised water level management strategies has generally been associated with the installation of new sluices and the increasing compartmentalisation of the wetland into distinct hydrological units. A model that can predict water level variations within these sub-units is therefore advantageous in the context of both present and future management.

To date, hydrological models applied on wet grassland have focussed on the dynamics of in-field water table variations (see Belmans, 1983, Youngs *et al.*, 1991, Armstrong ,1993), mainly due to the importance of phreatic conditions on wet grassland fauna and flora. Fewer models have sought to simulate the hydrology of the drainage network. Component ditches represent the focal point of the management strategies employed by both agricultural and nature conservation stakeholders to provide the water table conditions favoured by the variety of wet grassland stakeholders. The specific water level management targets for agriculture have been considered in Section 1.6.3 and for numerous wetland restoration schemes in the UK, water level targets included as management prescriptions have been reviewed in Section 1.7 and shown in Figure 1.15.

Where ditch water level modelling has been undertaken, to date this has generally been achieved by application of data intensive, distributed models such as MIKE-SHE (Al-Khundhairy *et al.*, 1997). Complex models of this kind rely on the intensive collection of data describing the physical properties of the area to which they are applied (Table 5.1). Data collection costs can therefore impose prohibitive demands where historical hydrological data have not been collected, and also require an experienced operator. However, complex models may not necessarily give better results than simpler ones (Beven and O'Connor, 1982). This is particularly the case in complex hydrological situations, such as those evident in wet grassland areas, where the accuracy of model output data is reliant on the preliminary stages of the modelling exercise, including the conceptualisation of the drainage system and the establishment of suitable boundary conditions. In wet grassland environments, this complexity is accentuated by the presence of looped channel systems, human interventions in the form of hydraulic structures, and the fact that the various components of the local hydrological system can be linked or isolated depending on local land use (Al-Khudhairy *et al.*, 1997). Field-scale management is strongly seasonal and there is normally little information regarding the precise management of sluice structures as landowners are responsible for them.

The overall objective of the model construction process described in this chapter was to develop a hydrological model with limited data requirements, capable of predicting ditch water levels in wet grasslands based on input data obtained by field survey or from published sources. Implicit in this objective is the examination of model data requirements and a means of identifying data collection priorities for wet grassland management authorities in the UK. This includes the type of data that should be collected, as well as their spatial and temporal resolution. The specific data requirements of hydrological modelling in wet grassland is considered in detail in Chapter 6, as are a series of assessments relating to the impacts of various water level management strategies and other scenarios on stakeholders of the Pevensey Levels wetland. Indeed, a complementary objective of the hydrological model was its ability to predict the effects of different water level management options on wetland stakeholders. Other scenarios such as climate change are also addressed, but a key focus is the use of the model as an interactive tool for the design of water level management strategies associated with WLMPs (Section 1.7.5) as the main strategy employed on the Pevensey Levels wetland to achieve 'wise use'. In this sense, the model is very much an extension of previous hydro-ecological research in wet grasslands (Newbold and Mountford, 1997, Gowing *et al.*, 1998). A critical difference however is the treatment of hydro-ecological data within an interactive, predictive tool for the on-screen assessment of the impacts or benefits of different water level management strategies on wetland stakeholders. The application of the hydro-ecological component of the model, as well as the data required by it, are described in Chapter 6. This chapter focuses only on the physical basis and mathematical process formulations associated with the hydrological model. This background provides the template for posterior scenario testing to address the variety of issues associated with the management of the Pevensey Levels wetland.

Model Component	Data requirements
Frame Parameters	• Ground surface elevation • Impermeable bed elevation • Distribution codes for meteorological source stations • Distribution codes for soil and vegetation types
Input data	• Meteorological and precipitation data
Initial conditions	• Phreatic surface levels • Overland and channel flow depths
Boundary conditions	• Surface flows or water levels at boundaries • Man-controlled channel flow diversions and discharges • Groundwater flows or potentials at boundaries • Groundwater pumping and recharge data
Interception Parameters	• Canopy drainage parameters • Canopy storage capacity (time varying) • Interception capacity coefficient • Rainfall rate
Evapotranspiration Parameters	• Ratio between AEt and PEt as a function of soil moisture tension • Canopy resistance • Aerodynamic resistance • Ground cover indices (time varying) • Leaf Area Index (time varying) • Root distribution with depth • Meteorological data
Overland and channel flow Parameters	• Strickler roughness coefficients; overland / river flow • Weir discharge coefficients • Channel geometry
Unsaturated zone Parameters	• Soil moisture tension/content relationship • Unsaturated K as a function of moisture
Saturated zone Parameters	• Porosity or specific yield • Saturated hydraulic conductivity

Table 5.1. Data requirements of the MIKE SHE model (After Singh, 1995).

5.2. A hydrological model for the prediction of water level fluctuations in ditches

5.2.1. MODEL STRUCTURE

Most of the data employed to construct and test the model described in this chapter has been previously described in Chapter 3. The hydrological model, called PINHEAD (**P**hysically-based **IN**tegrated **H**ydro-**E**cological **A**ssessment for **D**rainage ditch systems), employs a water balance approach to provide daily predictions of water level variations in the target ditch system. In doing so, PINHEAD represents a logical progression from the catchment-based water balance model described in Chapter 3, which provides predictions of monthly water availability at the catchment-scale. Indeed, catchment-scale studies have identified surface water storage in the field-scale ditch network as one of the components for which data are generally lacking (Section 3.1). The construction of PINHEAD is a response to this requirement, providing a means of quantifying this variable through time.

PINHEAD simulates the influence of the five key processes affecting ditch storage in wet grassland areas, including the seasonal variations in their magnitude. The hydrology of ditches in wet grassland areas is conceptualised in Figure 5.1. The main processes effecting level changes in drainage ditch systems are rainfall falling directly on the ditch water surface, evaporation from the water surface, runoff from surrounding fields, the interaction with the water table, and discharge through sluices. By treating sections of the ditch system between sluices as a closed reservoir, volumetric changes in ditch storage can be estimated as a daily succession of steady states where:

$$D_{S\,t} = D_{S\,t\text{-}1} + (P_V - E_V + R_V + G_V - Q_V) \qquad \text{(Equation 5.1)}$$

where $D_{S\,t}$ is ditch storage at 0900 GMT on day t, P_V and E_V are inputs and losses from the ditch surface by precipitation and evaporation respectively, R_V is the volume contributed by runoff, G_V represents the interaction between the water table and the ditch, and Q_V is sluice discharge at the downstream end of the ditch catchment. All data refer to the contributions or losses to the ditch system in the previous 24 hours and are expressed in m^3.

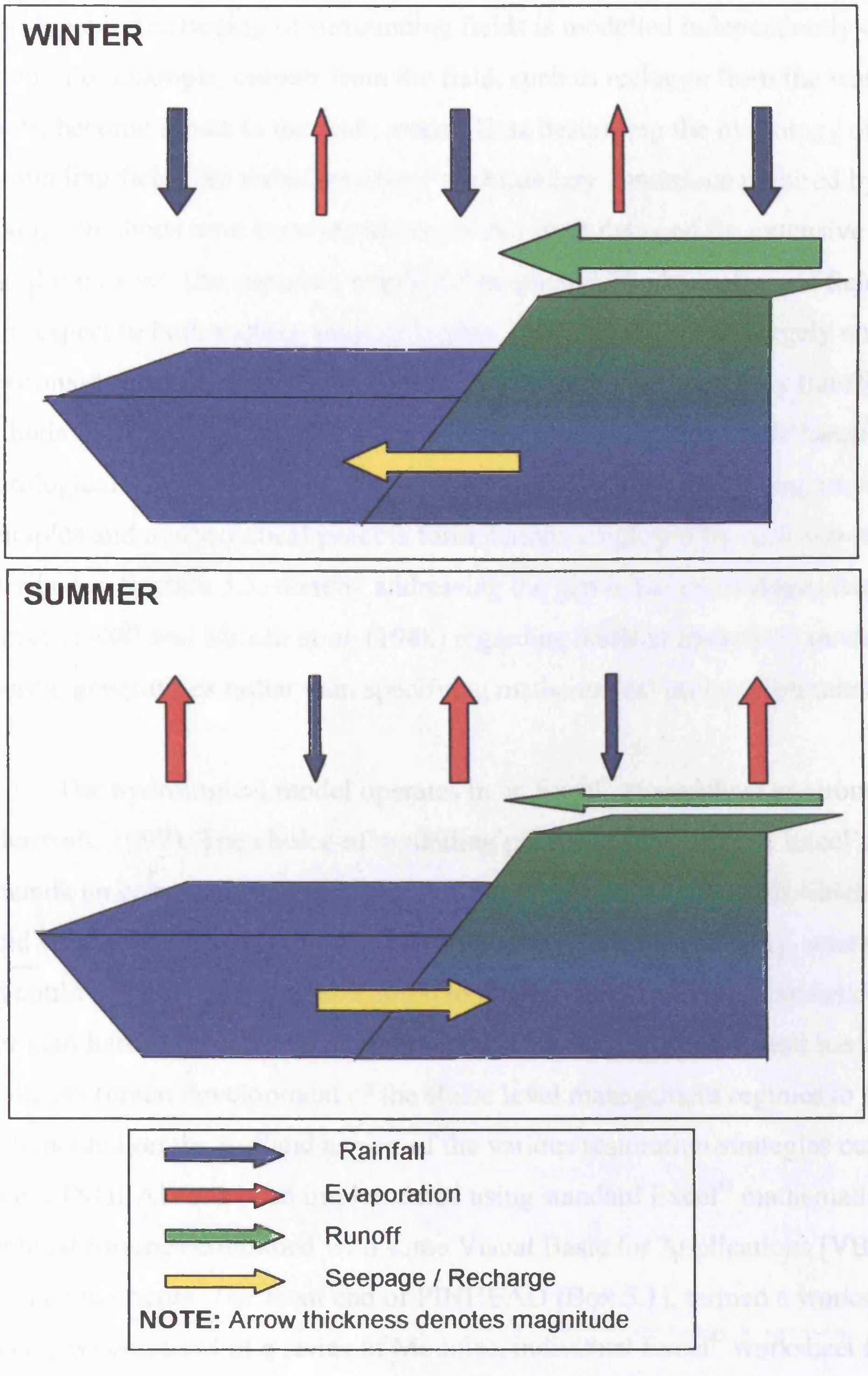
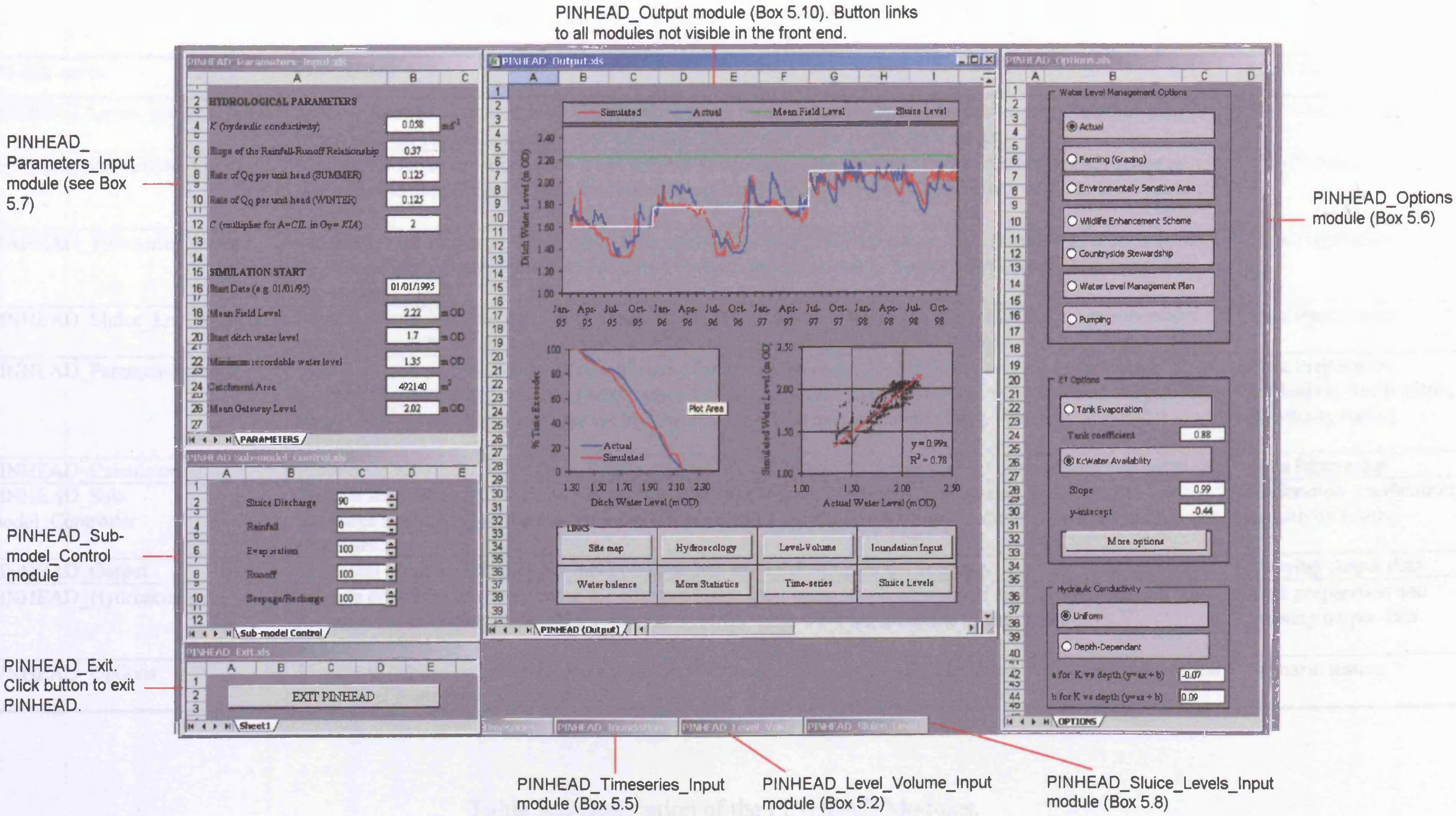


Figure 5.1. Conceptual representation of the hydrological processes effecting level changes in wet grassland ditch systems on a seasonal basis.

In PINHEAD, a sub-model represents each of the individual hydrological processes effecting volumetric changes in wet grassland ditch systems. The hydrological functioning of surrounding fields is modelled independently of the ditch system. For example, outputs from the field, such as recharge from the water table or runoff, become inputs to the ditch model. Data describing the hydrology of the surrounding fields are therefore one of the boundary conditions required by the model, although methods have been incorporated that limit the need for extensive data describing these. The approach employed to quantify the hydrology of field systems, with respect to both surface and sub-surface hydrology, is based largely on field-scale data considered in Chapter 3. This chapter also describes potentially transferable methods for the estimation of sluice discharge and runoff magnitude based on hydrological data routinely collected on the wetland. The functioning, physical principles and mathematical process formulations employed by each sub-model are described in Section 5.3, thereby addressing the gap in the knowledge identified by Duever (1988) and Mitsch *et al.* (1988) regarding wetland hydrology models that describe generalities rather than specifying mathematical process formulations.

The hydrological model operates in an Excel© spreadsheet environment (Microsoft, 1997). The choice of modelling platform was based on Excel's limited demands on computer hardware, wide availability and the ease with which the model could be implemented. An added advantage was that a user-friendly, interactive front-end could be designed for presentation to local stakeholders. Most stakeholders will have also had some previous contact with the software. The front end has been designed for the on-screen development of the sluice level management regimes to be implemented on the wetland as part of the various restoration strategies currently in place. PINHEAD has been implemented using standard Excel© mathematical and graphical routines combined with some Visual Basic for Applications [VBA](Microsoft, 1997) components. The front end of PINHEAD (Box 5.1), termed a workspace in Excel©, is composed of a series of Modules, individual Excel© worksheet files, which perform specific functions in data preparation, processing and output within the PINHEAD workspace. A descriptive summary of all the PINHEAD Modules is given in Table 5.2. Later sections refer more specifically to the functions individual Modules perform and elaborate on their use within the model implementation process.



Box 5.1. The front-end of PINHEAD: A guide to individual components. For an overview of the functions performed by each Module see Table 5.2.

Module name	Description	Used for
PINHEAD_Level_Volume_Input	Module for the calculation of the level-volume-area relationship of the target ditch system. Requires data describing the longitudinal and cross-sectional dimensions of the target ditch system.	Data Preparation
PINHEAD_Inundation_Input	Module for the calculation of the level-volume-area relationship associated with storage on field surfaces during inundation. Requires data describing inundation storage and extent at a variety of water level (from a DEM).	Data Preparation
PINHEAD_Timeseries_Input	Module for the incorporation of daily data required by the PINHEAD sub-models (rainfall, evaporation, soil moisture data). Also includes data for the calibration and validation of model output data (ditch water level time-series)	Data Preparation
PINHEAD_Sluice_Levels_Input	Module for the incorporation of the sluice level time series data required by the sluice discharge sub-model. Also has capacity for up to six simulated sluice level time series for scenario testing.	Data Preparation
PINHEAD_Parameters_Input	Module containing the key model parameters required to start a model run (start date, start water level, hydraulic conductivity, catchment area, mean field level, slope of the rainfall-runoff relationship, slope of the stage-discharge relationship of the controlling sluice for both summer and winter). Also used for model calibration.	Data Preparation, calibration, verification, sensitivity testing
PINHEAD_Calculator	Processing Module in PINHEAD, performing all calculations for the estimation of ditch water level variations	Data Processing
PINHEAD_Sub-model_Controller	Module controlling all the component sub-models in PINHEAD (rainfall, evaporation, runoff, sluice discharge and ground-surface water interactions). Can be used to evaluate the relative importance of different processes.	Calibration, verification, sensitivity testing
PINHEAD_Output	Graphical Module providing the model output data and evaluating model accuracy.	Viewing output data
PINHEAD_Hydroecology	Module providing data describing the effects of simulated water levels on wetland stakeholders/species. Requires the provision of data describing the specific water level requirements of the target stakeholder/species.	Data preparation and viewing output data
PINHEAD_Options	Module to control scenarios incorporated into the model. These are broadly classified as data requirement and water level management options.	Scenario testing

Table 5.2. Description of the PINHEAD Modules.

5.2.2. THE STUDY SITE

PINHEAD was developed for the main ditch system on the SWT reserve for the period 1st January 1995 - 31st December 1998. This period was coincident with that chosen for the catchment-scale water balance presented in Chapter 3. The catchment modelled was the larger of the two catchments previously defined on the nature reserve. In the analysis presented in Section 3.6, this catchment has been termed Field 2. The choice of location and period for model development and implementation was related mainly to the availability of data describing hydrological functioning. Specific details regarding the components of this field-scale monitoring network have been previously shown in Figure 3.32. Ditch water level and water table data have been collected at various locations on the reserve since early 1995. In Section 3.6.2, these data have been employed to establish the relationship between ditch and in-field water tables levels. In Section 5.5, ditch water level data collected on the SWT reserve are employed for the calibration and verification of model output data. On the SWT Reserve, data describing the sluice management regime during this period were also available, an important consideration in terms of the functioning of the sluice sub-model (Section 5.3.4).

Daily rainfall and evaporation estimates for the period January 1995 to January 1999 were obtained from the Environment Agency (EA) and were those data collected at the Horseye climate station previously employed in the catchment-based water balance presented in Chapter 3. To evaluate the importance of evaporation on overall field-scale hydrological functioning, the model was designed so that input evaporation data could be adjusted according to results presented in Section 4.7.3. In this way, results presented in Section 4.8 describing the influence of different evaporation estimates on the catchment water balance could be evaluated in the context of field scale systems and models. By choosing the SWT Reserve as the location for the development and implementation of the modelling approach described, the analysis of the evaporation data provided in Chapter 4 could be applied within the modelling framework with a large certainty regarding the applicability of the results in the local context.

5.2.3. CALCULATING LEVEL-VOLUME-AREA RELATIONSHIPS IN PINHEAD

The simulation of daily ditch water level variations based on Equation 5.1 requires a means of converting volumetric values of ditch storage (Ds_t) into a level equivalent. This is achieved using a level-volume-area relationship, a relationship that is also required by the rainfall and evaporation sub-models (Sections 5.3.1 and 5.3.2). Level-volume-area relationships are only suitable for hydrological systems where the water body is static and flow negligible for most of the time. As a result, they are ideal to model the hydrology of ditches in wet grasslands, where the flow of water occurs only when the marsh is being actively drained through a pump or sluice. They are also simple in conceptual terms, and their development requires limited data collection, both of which are important considerations in terms of the objectives of PINHEAD.

Level-volume-area relationships have been identified as an essential component of the hydrological data required for effective management in wetland areas (Hollis and Thompson, 1998; Hayashi and van der Kamp, 2000), and have received wide application in wetland hydrological studies to date (Reed, 1985; Sutcliffe and Parks, 1987, 1989; Thompson and Hollis, 1995). On the Pevensey Levels, these relationships have been previously employed to quantify storage in embanked channels and pumped sub-catchments on the wetland (Sections 3.4.3 and 3.5.3 respectively). The level-volume-area relationship is essentially the regression relationship between water level and both surface water storage and water surface area, and is established using empirical information describing the geometry of the channel system (Reed, 1985). In PINHEAD, the development of the level-volume-area relationship forms the first component of model implementation and can be expressed by a second-degree polynomial regression equation (Hayashi and van der Kamp, 2000) where:

$$\text{Area/Volume} = a\,\text{DWL}^2 + b\,\text{DWL} + c \qquad \text{(Equation 5.2)}$$

where DWL is ditch water level. The use of a second-degree polynomial curve allows for curvature in the relationship and can be generated by most statistical software, including Excel. In PINHEAD, the level-volume-area relationship takes the form of two regression relationships, one relating level to volume and one relating level to surface water area.

Values of a, b and c in Equation 5.2 are variables required by PINHEAD and can be obtained automatically within the PINHEAD_Level_Volume_Input Module (see Box 5.2). Data required for the calculation of the regression equations are:

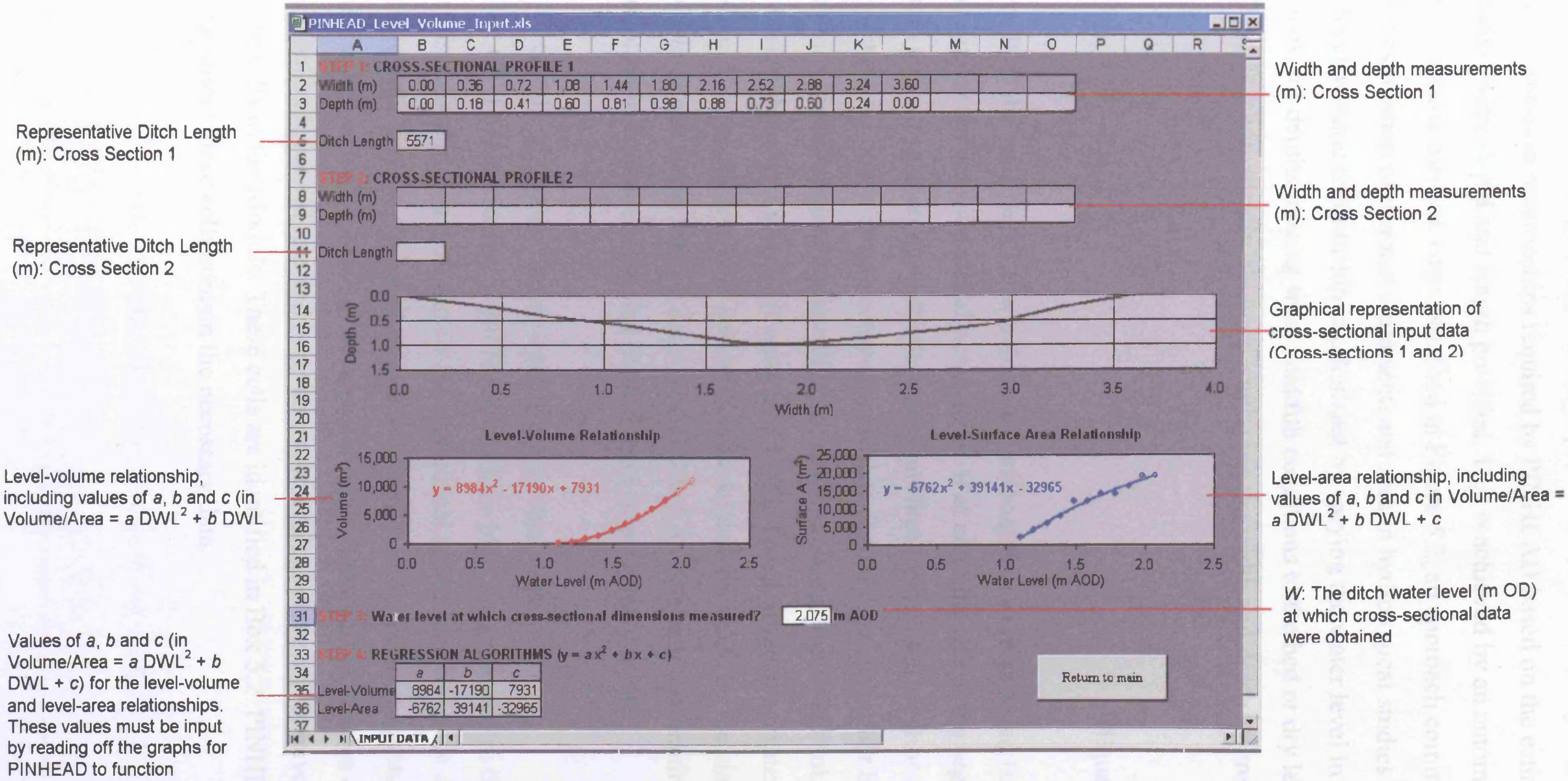
- the total length of the ditch system to be modelled (m),
- the cross-sectional dimensions of the ditch system, described by up to 15 depth and width measurement points taken from bank to bank (m), and
- the water level at which the cross-section employed was measured, termed W.

In the case presented, all elevation measurements were taken in metres above Ordnance Datum (m OD). For water level W, volumetric ditch storage ($D_{S\,t}$) can then be calculated by:

$$D_S = \mathrm{CSA_W}\, L \qquad \text{(Equation 5.3)}$$

where $\mathrm{CSA_W}$ is the cross sectional area (m^2) at water level W, and L is the representative length (m). Once the ditch catchment has been delineated, the value of L can be established from Ordnance Survey maps or more detailed information provided by the Internal Drainage Board (IDB). Methods for the delineation of ditch systems in wet grasslands are discussed in detail in the next section.

Data describing the cross-sectional dimensions and length of up to two ditch cross-sections in the target area can be provided. This is in response to previous analyses of ditches on the Pevensey Levels that have identified cross-sectional variation within individual sub-catchments as a feature of the drainage system. In pumped sub-catchments this is associated with the hierarchical design of the drainage system. For Type 1 ditches, differences can be ascribed to the date of construction (Section 1.6.1) or are associated with the creation of berms, a common practice in areas of conservation interest (Newbold *et al.*, 1989) where profiles may not coincide with the trapezoidal form typically evident in other areas. For example, a number of berms have been constructed on the SWT Reserve (Neil Fletcher, SWT Reserves Manager, Pers. Comm.) accounting for the variation in cross-sectional dimensions evident in Figure 3.24.



Box 5.2. The PINHEAD_Level_Volume_Area_Input Module.

The PINHEAD_Level_Volume_Input Module automatically calculates the level-volume-area relationships required by PINHEAD based on the estimates of cross-sectional width, depth and length provided. This is achieved by an automated version of the mid-section method, conceptualised in Figure 5.2, an approach commonly employed for the estimation of channel cross-sectional area in hydrological studies (Shaw, 1993). The level-volume relationship is calculated by varying the water level in each segment 10 times, for depths ranging from bankfull conditions to the bed or dry level ($\mathrm{DWL}_{\mathrm{Dry}}$). $\mathrm{DWL}_{\mathrm{Dry}}$ is automatically obtained employing the width and depth estimates provided by:

$$\mathrm{DWL}_{\mathrm{Dry}} = W - d_{\mathrm{Max}} \qquad \text{(Equation 5.4)}$$

where W is the water level at which cross-sectional data were gathered (m OD) and d_{Max} is the maximum cross-sectional water depth. If at any time one of the segments does not contain any water, then its width does not contribute to the cross-sectional surface area. For greatest accuracy, cross-sectional input data should represent either bankfull conditions, or be obtained by levelling the entire cross-section from bank to bank. This will ensure that water storage is replicated for the broadest range of water depths possible. If no cross-sectional data are available, data provided by the classification of Newbold *et al.* (1989) may be employed (Table 5.3), although this requires an assumption to be made regarding the bed level of the ditch system.

Graphical outputs are an integral component of the PINHEAD_Level_Volume_Input Module. Three plots are presented in the Module, which is shown in Box 5.2. The first is a graphical representation of the ditch cross-sectional data input to the Module. The level-volume and level-area plots are shown and include a regression equation of the type shown as Equation 5.2. Values of a, b and c shown on the level-volume and level-area relationships respectively have to be input to labelled cells in the Module. These cells are identified in Box 5.2. PINHEAD will not operate unless these cells contain the necessary data.

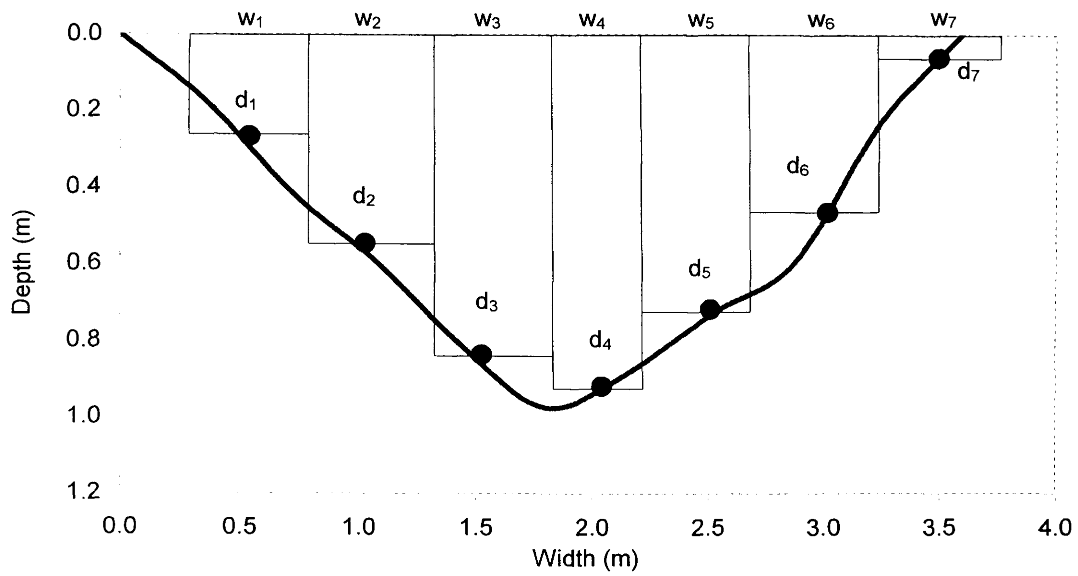


Figure 5.2. Graphical representation of the mid-section method employed to calculate cross-sectional and surface areas in the PINHEAD_Level_Volume_Input Module.

Type 1

Width (m)	0.00	0.50	1.00	1.50	2.00	2.50	3.00
Depth (m)	0.00	0.75	1.50	1.50	1.50	0.75	0.00

Type 2

Width (m)	0.00	2.00	4.00	6.00	8.00
Depth (m)	0.00	2.00	3.00	2.00	0.00

Type 3

Width (m)	0.00	2.00	4.00	6.00	8.00	10.00	12.00
Depth (m)	0.00	2.00	4.00	4.00	4.00	2.00	0.00

Type 4

Width (m)	0.00	2.50	5.00	7.50	10.00	12.50	15.00	17.50	20.00
Depth (m)	0.00	2.50	5.00	5.00	5.00	5.00	5.00	2.50	0.00

Table 5.3. Cross-sectional input data describing the dimensions of wet grassland ditch types according to Newbold *et al.* (1989).

For the development and implementation of the model on the SWT Reserve, the cross-sections employed were the mean of all the Type 1 ditches measured in the Field 2 catchment (Figure 3.24). By using the mean value, an assessment of the data requirements of the level-volume area relationship was included within the analysis. Width and depth measurements for the cross-section employed is shown in Table 5.4, and illustrated graphically in Figure 5.3. Cross-sectional measurements were taken at a water level of 2.075 m, which was therefore employed as the value of W in Equations 5.3 and 5.4. Based on the delineation of the Field 2 ditch system catchment (Section 5.2.4), the total length of the ditch system was 5561 m. For the Field 2 ditch catchment, ditch storage (D_S) could thus be expressed as:

$$D_S = 10{,}611\ DWL^2 - 20{,}303\ DWL + 9{,}367 \qquad \text{(Equation 5.5)}$$

and the water surface area ($D_{A\,t}$) by:

$$D_A = -7987\ DWL^2 + 46{,}230\ DWL - 38{,}936 \qquad \text{(Equation 5.6)}$$

In both cases, DWL is the ditch water level (m OD). A graphical representation of the level-volume and level-area relationships for the Field 2 catchment on the SWT Reserve is shown in Figure 5.4. Values of a, b and c for both relationships are shown in respective cells within the PINHEAD_Level_Volume_Input Module (Box 5.2).

5.2.4. DELINEATING DITCH SUB-CATCHMENTS ON THE SWT RESERVE

The calculation of level-volume-area relationships is reliant on the identification and delineation of hydrologically-discrete sub-catchments within the intricate networks of channels that characterise wet grassland landscapes. Unlike highland catchments where the topographic catchment boundary is formed exclusively by the terrain, catchment boundaries in low-lying areas are often much more difficult to determine (Marshall, 1989). Nevertheless, a number of simple methods for the delineation of ditch catchments were established during the development of PINHEAD. Road or rail embankments, which are raised above the surrounding low-lying land, are frequently found to be catchment boundaries (Beran, 1987, Marshall, 1989). These features can be observed on 1:25,000 O.S. maps and therefore have wide applicability for similar approaches elsewhere.

Width (m)	0.00	0.36	0.72	1.08	1.44	1.80	2.16	2.52	2.88	3.24	3.60
Depth (m)	0.00	0.18	0.41	0.60	0.81	0.98	0.88	0.73	0.60	0.24	0.00

Table 5.4. Cross-sectional input data employed for the establishment of the level-volume-area relationships on the SWT Reserve. Data shown are the mean cross-sectional dimensions of all Type 1 ditches measured in the Field 2 catchment.

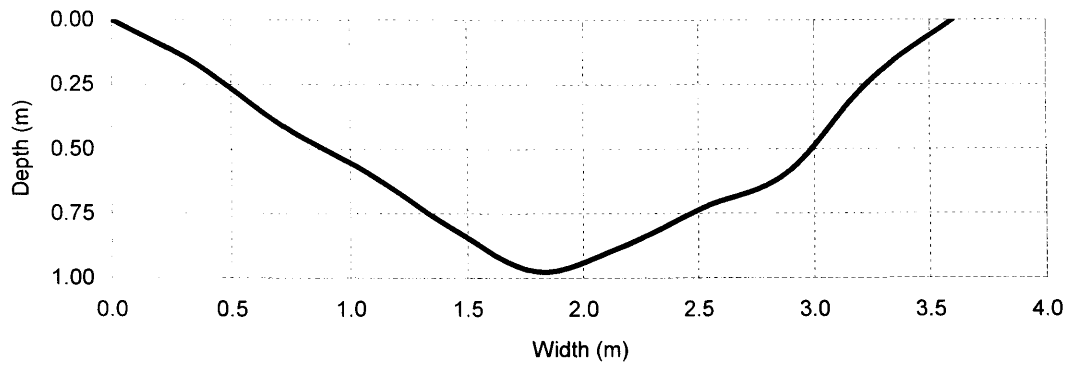


Figure 5.3. Ditch cross-sections used to estimate level-volume-area relationships on the SWT Reserve.

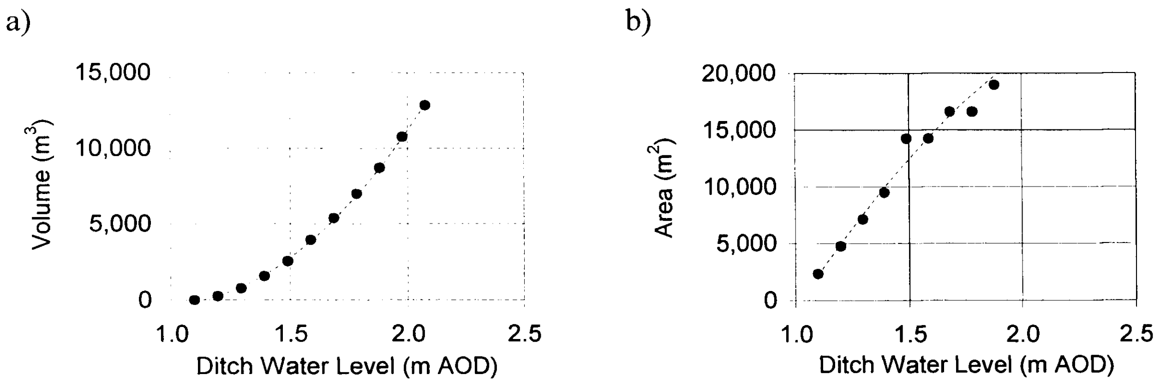


Figure 5.4. (a) Level-volume and (b) level-area relationships for the Field 2 ditch system on the SWT reserve.

The general design principles of the drainage network can also be used for effective catchment delineation. Water level control structures are frequently found at the junction of different ditch types. In the application of PINHEAD, these can be used as catchment delineators because the simulation of sluice discharge is an integral component of the hydrological model (Section 5.3.4). The IDB of a wet grassland should keep records of all the structures over which they have responsibility, including their location, and possibly their dimensions. The boundary of the area dependant on a water level control structure is defined by blocked ends or bunds, which are constructed by the landowner to isolate an area and allow flexible water level management. On the SWT Reserve, blocked ends are also commonly gateways.

In wet grassland sites where water levels are managed for nature conservation, it may be easier to delineate ditch catchments than in agriculturally-dominated areas. This is because hydrological isolation from surrounding land is commonly one of the first measures implemented when establishing a wetland nature reserve (RSPB *et al.*, 1997). On the SWT Reserve for example, maps provided by the SWT marked the exact location of the three blocked ends and one sluice that were installed during the instatement of the nature reserve. Nevertheless, catchment delineation on the SWT reserve clearly supported the need for field verification of the location and status of catchment delineators. The structures for water level management were varied and not always visible on the field surface. On the SWT reserve these included piped crossovers linking ditches (Figure 5.5) which only became evident during the dry summers of 1995 and 1996. Field verification also identified that many of these were blocked by the characteristically clay-silt substrate and did in fact act as blocked ends.

Consecutive field visits allowed the identification of two distinct catchments within the reserve: a small catchment to the north and a larger catchment to the south. The southern catchment was that termed Field 2 in Chapter 3, and the northern catchment was Field 3. The boundaries of each catchment are shown in Figure 5.5. Because a number of fields on the reserve were connected to both ditch systems, the precise boundary between the two catchments was defined by the distribution of grips on the Reserve, digitized from aerial photographs, and shown in Figure 5.5. Parts of individual fields contributing to either the Field 2 or Field 3 ditch system were identified by applying traditional catchment delineation techniques to grip networks and topographical data for individual fields.

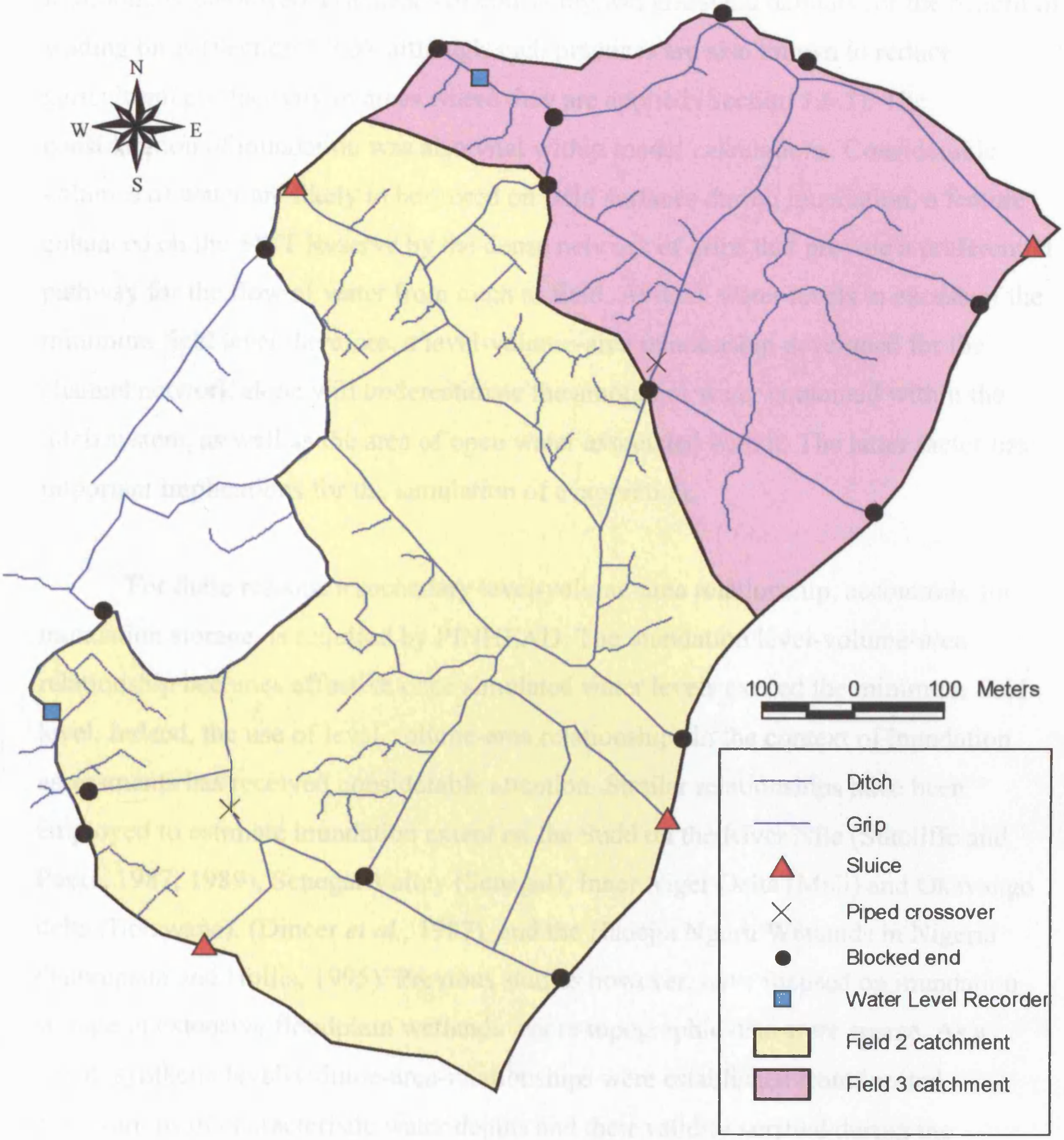


Figure 5.5. Location of catchment delineators on the SWT reserve, and the catchment boundaries defined from them.

5.2.5. THE PINHEAD_INUNDATION_INPUT MODULE

An important component of the objectives of the construction of PINHEAD was its ability to estimate the extent and duration of inundation events under different management, and climatic, scenarios. The inundation of field surfaces has been traditionally employed as a means of enhancing wet grassland habitats for the benefit of wading birds (Section 1.7.3), although such practices are also known to reduce agricultural productivity in areas where they are applied (Section 1.6.5). The consideration of inundation was also vital within model calculations. Considerable volumes of water are likely to be stored on field surfaces during inundation, a feature enhanced on the SWT Reserve by the dense network of grips that provide a preferential pathway for the flow of water from ditch to field. At ditch water levels in excess of the minimum field level therefore, a level-volume-area relationship developed for the channel network alone will underestimate the amount of water contained within the ditch system, as well as the area of open water associated with it. The latter factor has important implications for the simulation of evaporation.

For these reasons a secondary level-volume-area relationship, accounting for inundation storage, is required by PINHEAD. The inundation level-volume-area relationship becomes effective once simulated water levels exceed the minimum field level. Indeed, the use of level-volume-area relationships in the context of inundation assessments has received considerable attention. Similar relationships have been employed to estimate inundation extent on the Sudd on the River Nile (Sutcliffe and Parks, 1987, 1989), Senegal Valley (Senegal), Inner Niger Delta (Mali) and Okavango delta (Botswana), (Dincer *et al.*, 1987), and the Hadejia Nguru Wetlands in Nigeria (Thompson and Hollis, 1995). Previous studies however, have focused on inundation storage in extensive floodplain wetlands where topographic data were sparse. As a result, synthetic level-volume-area-relationships were established from limited observations of characteristic water depths and their validity verified during the calibration of models employing these data. In contrast, for the area to which PINHEAD was applied, detailed topographical data based on a 30m grid sampling strategy were available. These data were employed to derive a detailed Digital Elevation Model (DEM) of the SWT Reserve that could be used to evaluate inundation storage and extent in an area less than 1 km^2 in extent.

The development of the inundation level-volume-area relationship probably represents the most data intensive component of PINHEAD. These data therefore represent the greatest barrier to the transferability of the model to areas other than the Pevensey Levels wetland. The establishment of level-volume-area relationships for inundation requires detailed topographical data describing the target catchment. Data required include the inundation threshold water level (in m OD), and the volumetric storage and open water area associated with up to six water levels greater than the inundation threshold water level. Ideally, these six water levels will be representative of the entire range of water levels between the minimum and maximum field levels in the target catchment. For the SWT Reserve, inundation storage and extent at a range of water levels could be calculated from a grid-based topographical survey conducted by ADAS in 1993 (Armstrong, 1998). A secondary 'layer' of this survey employed a Geographical Positioning System (GPS) to map inundation extent at different locations during the winter of 1998, when much of the reserve was flooded. Water depth at each location was measured using a metre rule and, combined with data describing the water level recorded at the Field 2 water level recorder during the time of the survey, employed to calculate the elevation of each survey point in m OD. This secondary survey focused mainly on the grip system, the topography of which could not be accounted for by the ADAS grid-based survey.

Figure 5.6 shows the Digital Elevation Model (DEM) of the SWT Reserve obtained by combining the ADAS and GPS surveys. The survey points are also shown. The DEM was generated using the 3D Analyst extension in ArcView (Environmental Systems Research Institute, 1996) that is capable of interpolating a grid from x, y and z coordinates. Software packages such as Surfer (Golden Software, 1993) may also be used, although ArcView is preferred because data can be processed so that areas outside catchment boundaries are not incorporated within the grid. The minimum field level on the SWT Reserve, the level at which inundation of field surfaces was initiated, was 2.00m OD. For water levels in excess of this threshold, inundation storage and extent were calculated in ArcView, firstly by converting the processed grid to a .TIN (Triangulated Irregular Network) file, and secondly by applying the Volume Area Statistics function of the 3D Analyst extension. Table 5.5 shows inundation storage and extent for various water levels in the Field Two and Three ditch systems

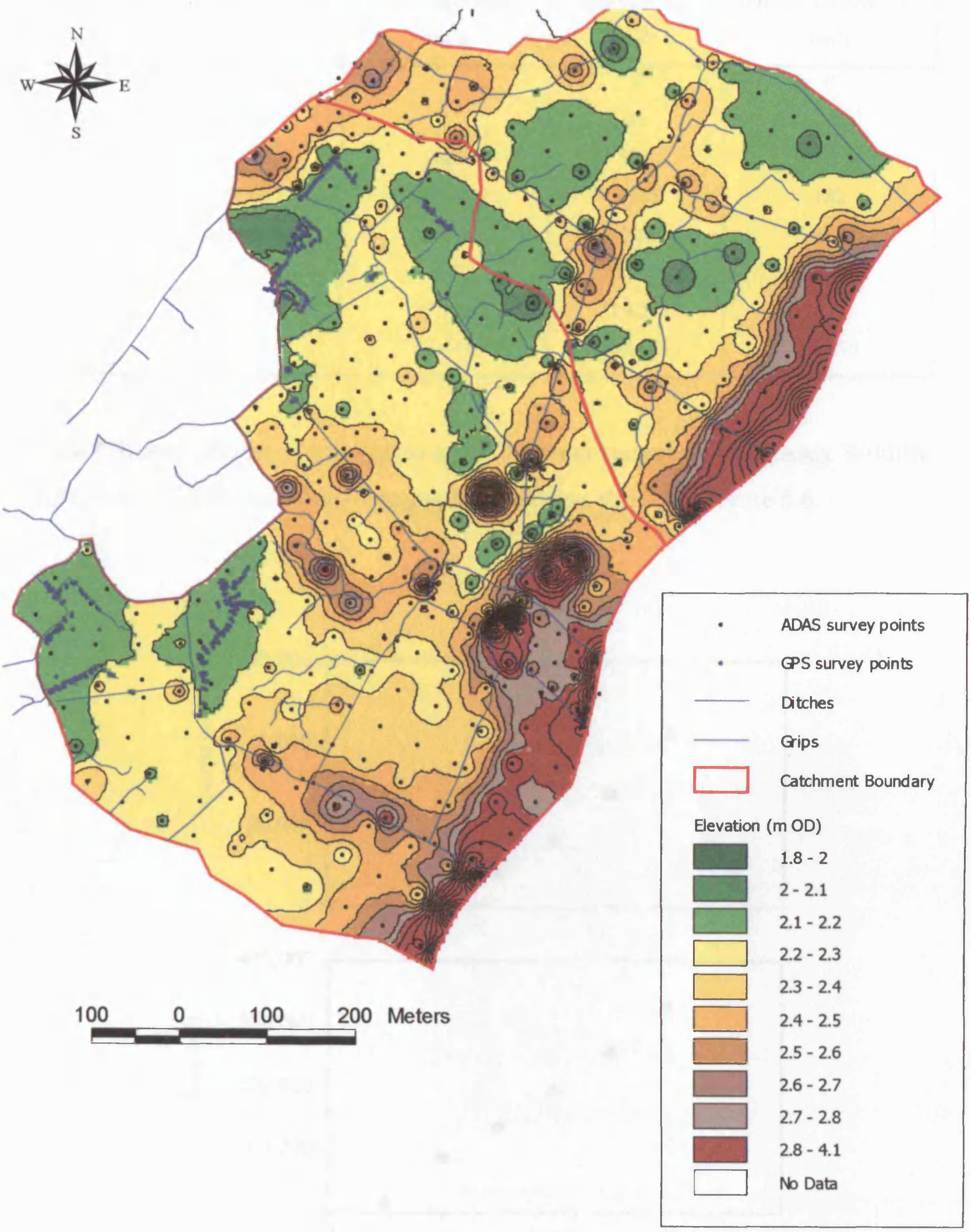


Figure 5.6. Digital Elevation Model (DEM) of the two ditch catchments on the SWT Reserve, including topographical survey points.

	Field 2 catchment		Field 3 catchment	
Water Level (m OD)	Inundation Area (m^2)	Inundation Storage (m^3)	Inundation Area (m^2)	Inundation Storage (m^3)
1.95	0	0	0	0
2.00	129	1	41	0.5
2.05	11,717	142	976	18
2.10	86,677	2,752	7,535	182
2.15	135,636	8,316	27,247	985
2.20	191,279	16,493	65,174	3,262
2.25	252,836	27,533	119,336	7,872
2.30	327,851	42,046	157,935	14,855

Table 5.5. Areal extent of inundation and inundation storage on the Sussex Wildlife Trust Reserve calculated from topographical surveys shown in Figure 5.6.

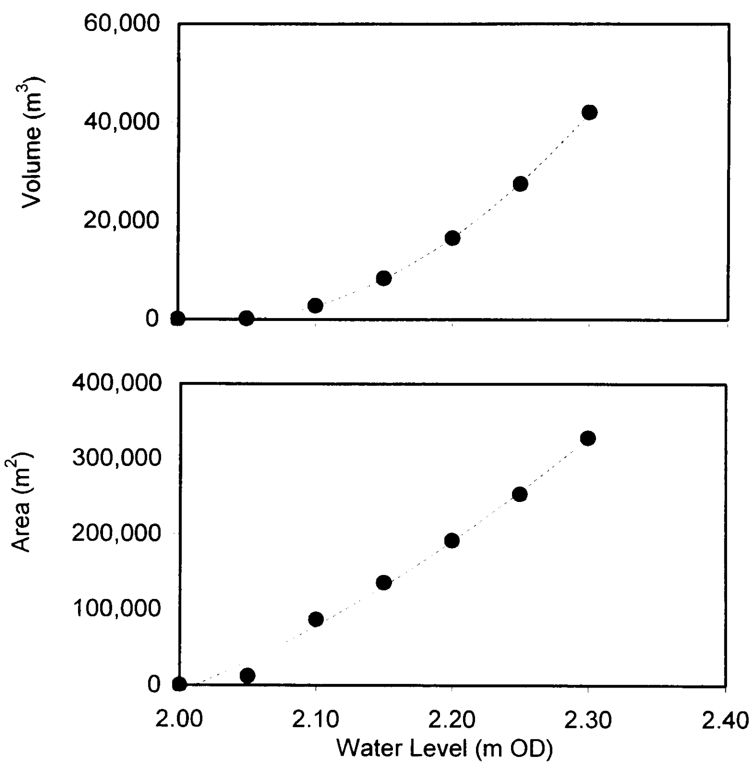


Figure 5.7. Level-volume and level-area relationships for inundation on the Field Two ditch system on the SWT reserve.

The level-volume-area relationships for inundation required by PINHEAD can be calculated automatically in the PINHEAD_Inundation_Input Module. Data describing inundation storage and extent, obtained from a DEM of the target catchment, are input to the cells specified in Box 5.3. These data are automatically combined with estimates of volumetric storage and surface water extent in the ditch system. At water levels in excess of the inundation threshold therefore, the inundation level-volume-area relationship provides an indication of both the water stored in the ditches and field surfaces, as well as the extent of the water surface. A simple Excel© IF statement controls the application of either the level-volume-area relationship for the ditch and inundation, or for the ditch only. The statement is linked to the threshold inundation water level, which has to be input for the model to work correctly (Box 5.3). If water levels during a given time-step exceed this threshold, the level-volume-are relationship associated with the ditch and inundation is employed. If water levels remain below this threshold, the level-volume-area relationship for the ditch only is applied.

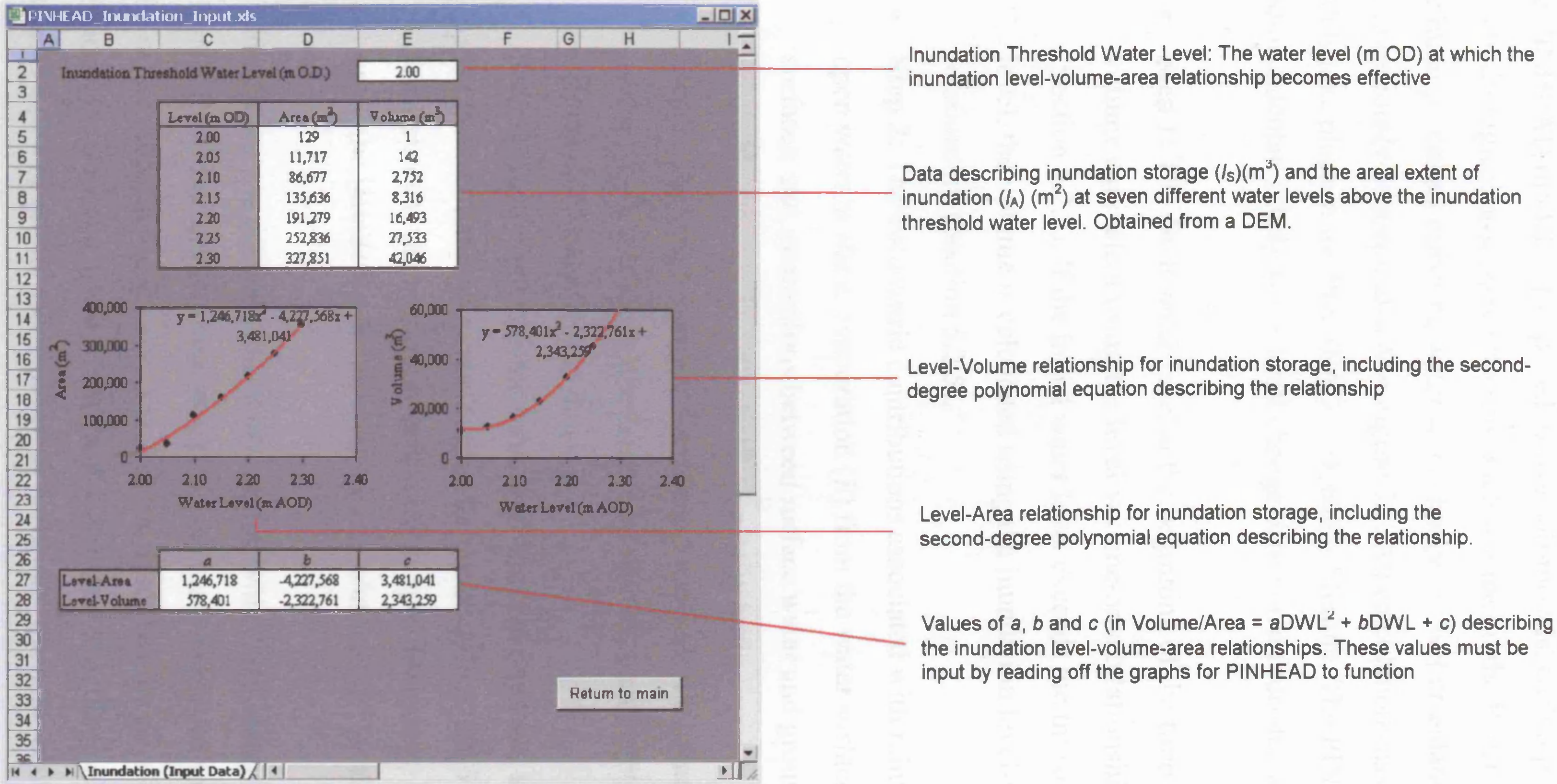
The level-volume and level-area relationships for the Field 2 catchment have been previously given as Equations 5.5 and 5.6 respectively. For the Field 2 catchment, the level-volume-area relationships for inundation are illustrated graphically in Figure 5.7. As in the case of the Level_Volume_Input Module, the Inundation_Input module in PINHEAD provides a graph describing the relationships between ditch water level, volume and area. For consistency, a second-degree polynomial regression equation is employed to quantify the level-volume-area relationship for inundation. For the Field 2 catchment, and based on Figure 5.7, inundation storage (I_S) (m^3) was given by:

$$I_S = 578{,}401\ DWL^2 - 2{,}322{,}761\ DWL + 2{,}343{,}259 \qquad \text{(Equation 5.7)}$$

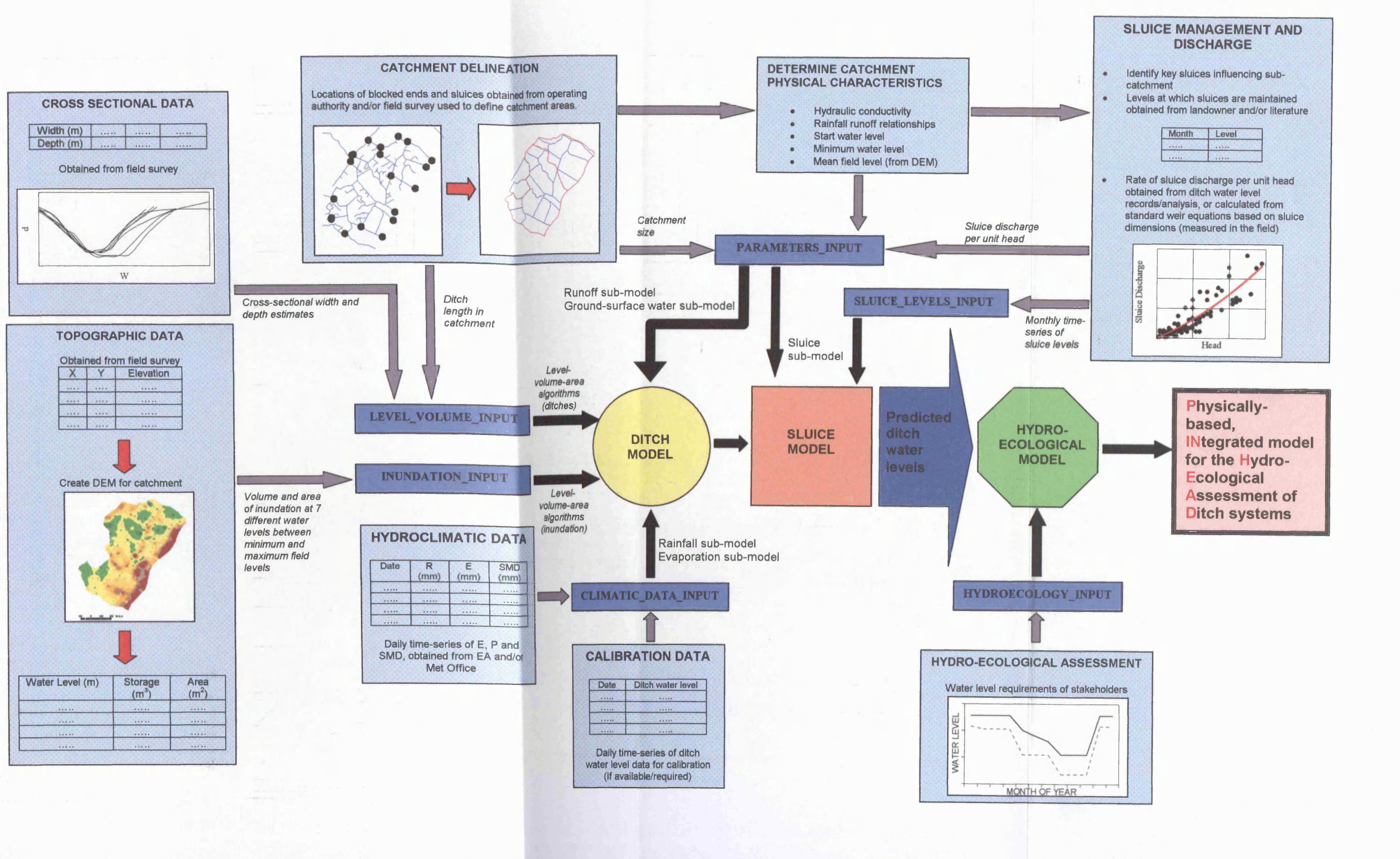
where DWL is ditch water level in m OD. Inundation extent (I_A) could be estimated by:

$$I_A = 1{,}246{,}718\ DWL^2 - 4{,}227{,}568\ DWL + 3{,}481{,}041 \qquad \text{(Equation 5.8)}$$

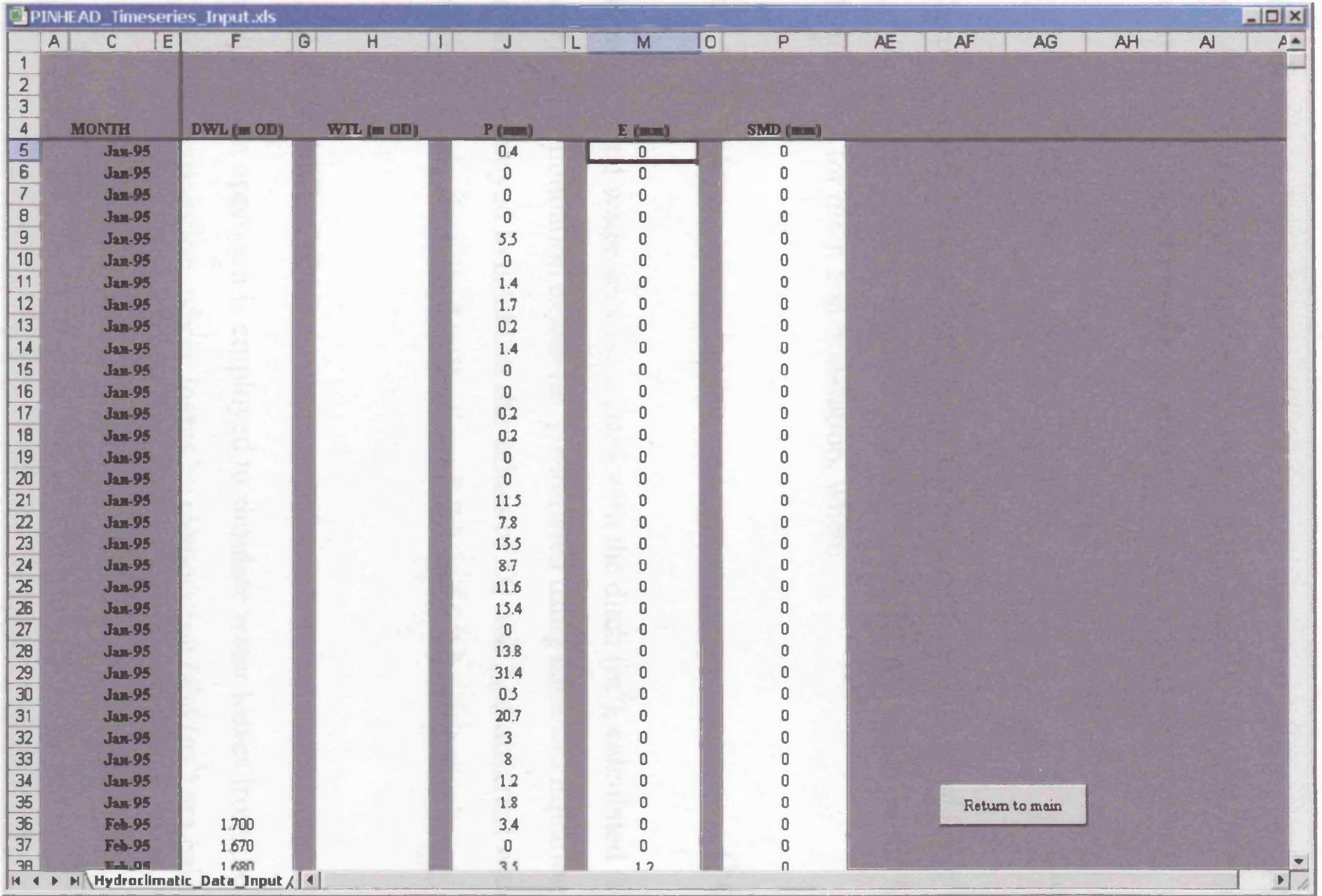
These relationships were effective at water levels in excess of 2.00m OD, the inundation threshold water level based on the DEM for the Field 2 catchment (Figure 5.6). As in the case of the PINHEAD_Level_Volume_Input Module, values of a, b and c for the inundation level-area and level-volume relationships have to be input to specified cells in the PINHEAD_Inundation_Input Module (Box 5.3) for the model to work correctly.



Box 5.3. The PINHEAD_Inundation_Input Module.



Box 5.4. Data requirements and linkages between different Modules in PINHEAD.



PINHEAD_Timeseries_Input.xls

MONTH	DWL (m OD)	WTL (m OD)	P (mm)	E (mm)	SMD (mm)
Jan-95			0.4	0	0
Jan-95			0	0	0
Jan-95			0	0	0
Jan-95			0	0	0
Jan-95			5.5	0	0
Jan-95			0	0	0
Jan-95			1.4	0	0
Jan-95			1.7	0	0
Jan-95			0.2	0	0
Jan-95			1.4	0	0
Jan-95			0	0	0
Jan-95			0	0	0
Jan-95			0.2	0	0
Jan-95			0.2	0	0
Jan-95			0	0	0
Jan-95			0	0	0
Jan-95			11.5	0	0
Jan-95			7.8	0	0
Jan-95			15.5	0	0
Jan-95			8.7	0	0
Jan-95			11.6	0	0
Jan-95			15.4	0	0
Jan-95			0	0	0
Jan-95			13.8	0	0
Jan-95			31.4	0	0
Jan-95			0.5	0	0
Jan-95			20.7	0	0
Jan-95			3	0	0
Jan-95			8	0	0
Jan-95			1.2	0	0
Jan-95			1.8	0	0
Feb-95	1.700		3.4	0	0
Feb-95	1.670		0	0	0
Feb-95	1.680		3.5	1.2	0

Box 5.5. The PINHEAD_Timeseries_Input Module.

5.3.1. RAINFALL

Daily rainfall data are employed to estimate inputs to the ditch system from water falling directly onto the open water surface. A daily rainfall time series for the period modelled are required for this purpose, and are input to the column labelled 'Rainfall' in the PINHEAD_Timeseries_ Input Module, shown in Box 5.5. For the SWT Reserve, daily Horseye rainfall data employed in previous water balance assessments were considered suitable due to the proximity of the gauge to the site. In PINHEAD, the volumetric contributions of rainfall, P_V (m^3), are then calculated by

$$P_V = [P / 1000]\, T_{SWA} \qquad \text{(Equation 5.10)}$$

where P is rainfall (mm) and T_{SWA},is the total surface water area (in m^2). At each time-step, T_{SWA} is calculated automatically within PINHEAD based on the level-area relationships for ditch and inundation, where

$$T_{SWA} = D_A + I_A \qquad \text{(Equation 5.11)}$$

where D_A is the water area associated with the ditch (m^2), calculated from Equation 5.6, and I_A is the inundation extent (m^2), estimated using equation Equation 5.8. The methods employed to establish the values of D_A and I_A at different water levels have been previously described in Sections 5.2.3 and 5.2.5 respectively.

5.3.2. EVAPORATION

An equivalent approach is employed to simulate water losses from the ditch water surface by evaporation, where losses by evaporation (E_V) (m^3) are calculated by:

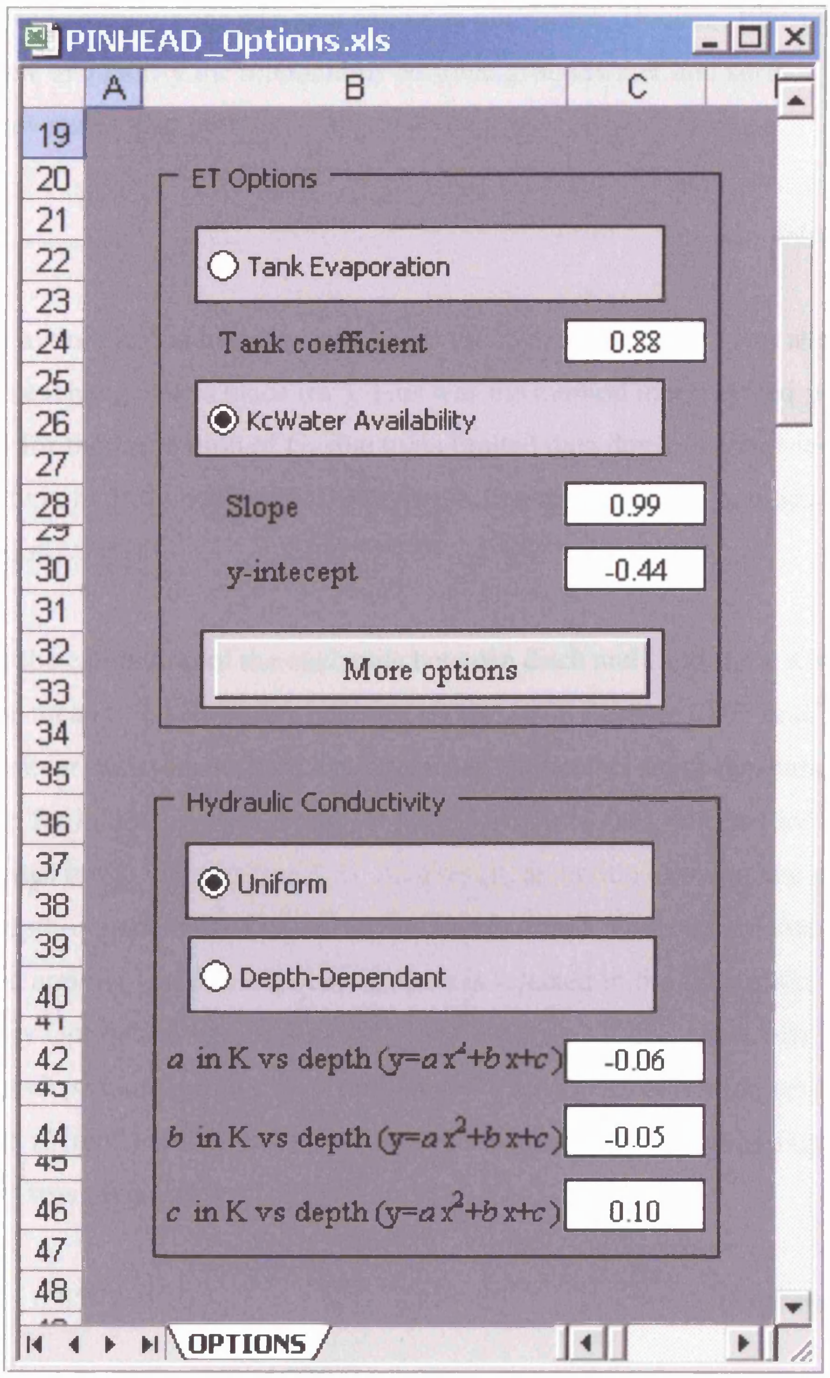
$$E_V = [E / 1000]\, T_{SWA} \qquad \text{(Equation 5.12)}$$

As in the case of rainfall, a daily evaporation time series for the period modelled is required. This time series is input to the column labelled 'Evaporation' in the PINHEAD_Timeseries_ Input Module in the main model screen (Box 5.5). For the SWT Reserve, data employed were Horseye tank evaporation estimates that, as previously stated, were the only continuous evaporation estimate available for the entire study period. To incorporate the results described in Chapter 4, two options have been incorporated to the PINHEAD model to calculate evaporative losses. The first option enables the application of standard constants such as tank coefficients (Section 4.3) to input evaporation data. Secondly, evaporation data can be adjusted according to the method described in Section 4.7.3, where actual evaporative loss is estimated from the reference evaporation rate, assumed to be that input to the PINHEAD_Timeseries_ Input Module and the ditch water level.

The specific method applied within model calculations is selected in the Evaporation frame of the PINHEAD_Options Module (Box 5.6). The Evaporation frame of the PINHEAD_Options Module also allows the evaluation of the influence of using different evaporation estimates on the accuracy of output ditch water level data, an issue considered in detail in Chapter 6. For simulations associated with the use of a tank coefficient, a constant of 0.88 was employed on the SWT Reserve as applicable to 'coastal, dyked wetlands' (Kadlec, 1989). The use of the $Kc_{\text{Water Availability}}$ option based on the results presented in Section 4.7.3, the default option applied in PINHEAD, requires values of a and b describing the linear relationship between daily ditch water levels (in m below field level) and $Kc_{\text{Water Availability}}$. For the Horseye evaporation tank that provided the input evaporation data, values of a and b describing that relationship were obtained from Table 4.10.

5.3.3. SURFACE-GROUNDWATER INTERACTIONS

The importance of the interactions between water table and ditch in wet grasslands have been previously recognised in water table models applied to wet grasslands (Belmans, 1983, Youngs, 1991, Armstrong, 1993). All these models require a measure of the ditch water level at each time step. Hydrological monitoring on wet grasslands has allowed the identification of the distinctly seasonal behaviour of the hydraulic gradient between ditch and field (Section 1.6.4), a seasonal variation that has been found to be applicable on the Pevensey Levels wetland (Figure 3.37). Numerous hydraulic solutions to quantify the interactions between surface and shallow groundwater have been proposed. In agricultural drainage studies, the most common approach for the estimation of G_V has been Hooghoudt's formula (Feddes, 1988; Smedema and Rycroft, 1983). However, an important limitation of the applicability of this method on the Pevensey Levels was the difficulty of identifying an impermeable layer beneath the ditch system, a parameter required for the calculation of surface-groundwater interactions at each time step.



Box 5.6. PINHEAD_Options Module showing available options incorporated to calculate model evaporation and ground-surface water interactions.

Indeed, piezometer data reviewed in Section 3.6.4 provide firm evidence that water moves more rapidly through the underlying peat than through overlying clays complicating the application of Hooghoudt's formula in the local context. In a study estimating the contributions of a peat soil to an open ditch, Boelter (1968) has employed Darcy's Law to quantify the interactions between groundwater and surface water. Darcy's Law states that:

$$G_V = KIA \qquad \text{(Equation 5.13)}$$

where K is hydraulic conductivity (md^{-1}), I is the hydraulic gradient (m) and A is area over which exchange takes place (m^2). This was the method incorporated within PINHEAD for the estimation of G_V due to its limited data demands, its wide applicability in hydrological studies, and the limited theoretical difficulties involved in its application.

For the estimation of the exchange between ditch and field (G_V), K was initially taken as the mean of all estimates gathered on the SWT Reserve (0.057md^{-1}) (Section 3.6.2). However, analysis of these data identified a potential depth-dependant relationship for hydraulic conductivity (Figure 3.36), a feature reported for other wet grassland sites in the UK (Section 1.5). As a result, an option allowing the use of either single or depth-dependant values of K to estimate G_V was incorporated into PINHEAD. The method applied within model calculations is selected in the 'Hydraulic Conductivity Options' frame in the PINHEAD_Options Module (Box 5.6). For the use of the depth-dependant option, values of a and b for a linear regression between level (m OD) and K (md^{-1}) are required. For the SWT Reserve, and based on Figure 3.36, this relationship was given by

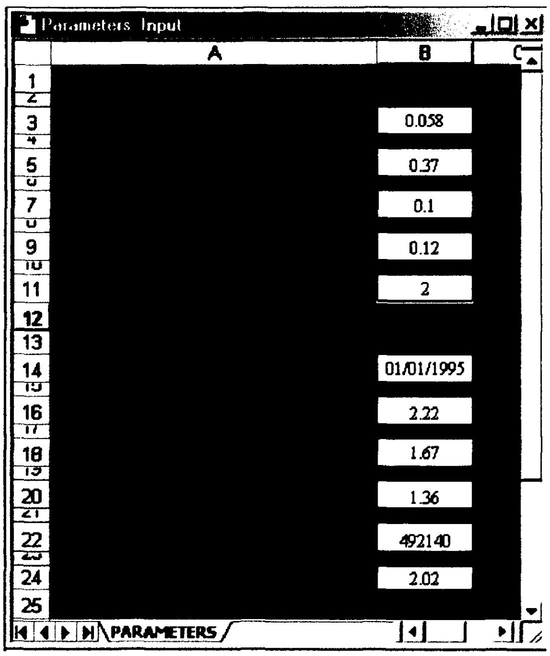
$$K= -0.06d^2 - 0.05d + 0.10 \qquad \text{(Equation 5.14)}$$

where d is depth below the mean field level (m), a parameter that is required by the PINHEAD_Parameters_Input Module (Box 5.7). For the SWT Reserve, this value was accordant with an elevation of 2.22m OD based on the DEM of the site. For the use of a single value of K for the estimation of G_V, a value for this parameter need to be input to the PINHEAD_Parameters_Input Module (Box 5.7). Ideally, this estimate is obtained from field measurement, although where no other data are available the operator can use values obtained from Table 1.8 describing the hydraulic conductivities of a range of soils in other wet grassland sites in the UK.

In PINHEAD calculations, the hydraulic gradient between field and ditch (I) are represented by the difference between the ditch water level and the water table level. Section 3.5.1 has illustrated that, for the Field 2 ditch system, this trend can be explained in terms of the Soil Moisture Deficit [SMD] (Figure 3.49). This relationship offered the opportunity to quantify I at each time step based on data routinely collected on the wetland by:

$$I = -0.0022\text{SMD} + 0.1617 \qquad \text{(Equation 5.15)}$$

where SMD is Soil Moisture Deficit (mm) relating to actual evaporation, obtained from the Meteorological Office MORECS system (Hough *et al.*, 1997).



Box 5.7. The PINHEAD_Parameters_Input Module.

At each model time-step, the value of A in Equation 5.13 is calculated as a function of the hydraulic gradient. This assumption is based on previous work by Boelter (1968). This work indicated that the largest exchange of water took place in the region where the hydraulic gradient was greatest, an aspect of groundwater-surface water interactions also been noted by Miles (1980). In keeping with the general objectives of the construction of PINHEAD therefore, a simple approach was employed for the calculation of A at each time step, where:

$$A = C\,I\,L \qquad \text{(Equation 5.16)}$$

L is the total ditch length (m) and I is the hydraulic gradient (m), determined by application of Equation 5.16. The use of the ditch length data (L) incorporates the assumption that groundwater-surface water interactions take place over the entire length of the ditch system. C is a constant included in the PINHEAD_Parameters_Input Module (Box 5.7) that can be used for the optimisation of ditch water level output data. Based on previous work by Boelter (1968) and Miles (1980), the value of C should be at least 2 because the groundwater-surface water interactions occur on both banks of the ditch system.

5.3.4. SLUICE DISCHARGE

5.3.4.1. Weir discharge equations to quantify sluice discharge

The development of a method for the estimation of sluice discharge was an integral, and data-intensive, component of the development of PINHEAD. Modelling sluice discharge was pre-requisite as there are over 250 structures for water level control on the Pevensey Levels wetland. In recent times, the installation of new structures, or the re-profiling of existing ones, has also been a central activity associated with the implementation of Water Level Management Plan's (Section 2.8.2).

However, few methods for quantifying discharge through sluices were initially identified in the literature. In areas where such data have been previously required, a common approach has been the installation of v-notch weirs in controlling structures (K. Gilman, Hydrologist, Pers Comm.; D. Marshall, CEH Wallingford, Pers. Comm.). Another potential approach is the implementation of equations describing discharge through other weir types that are analogous in form to the target sluice (Kadlec, 1983). On the Pevensey Levels, penning-board sluices are by far the most common water control structure (see Section 2.4.6; Blackmore, 1993), accounting for 77 % of all structures on the wetland and 82% of all structures present on Type 1 ditches (Section 2.4.6). In the Field 2 catchment, all sluices were of the penning-board type. A major advantage in applying weir-discharge equations to penning-board structures is that in form at least, they are analogous to rectangular weirs with side contractions. Engineering texts provide numerous empirical methods for the estimation of flow through rectangular weirs (*e.g.* British Standards Institution, 1982, 1990, French, 1994). Three common methods are reviewed in Table 5.6. The data required by all the methods considered are the dimensions of the weir, including b, the width of the weir, B, the width of the approach channel, P, the height of the weir crest relative to the channel bed, and h, the hydraulic head. In the case of water level control structures, b can be taken as the width of the sluice penning board. B is the width of the ditch upstream of the sluice and P is the distance between the penning board level and the ditch bed level. The hydraulic head (h) can be assumed to be equivalent to the difference between actual water level and the sluice level. By applying these equations at increasing values of h, a stage-discharge relationship to be implemented in the model was developed.

Water levels in the Field 2 ditch system are controlled mainly by sluice P26, labelled in Figure 5.5 and shown in Figure 5.8. The dimensions of this sluice P26 could be summarised as b=1.4m, B= 3.5m and P = 1m. These dimensions were found to be characteristic of all penning-board structures on the SWT Reserve, as well as of many structures elsewhere on the wetland, supporting the potential transferability of the method to other ditch catchments on the wetland. All three methods identified in Table 5.6 were tested for incorporation into PINHEAD. For sluice P26, stage-discharge relationships developed based on the methods shown in Table 5.6 are shown in Figure 5.9. Of all the stage-discharge relationships developed, the greatest range in values of discharge with h was provided by the Kinksvarter and Carter (1957) method. The smallest range in discharge estimates was obtained from the Swiss Engineers Association (SIA) (1924) method. Similar results were provided by the Hamilton-Smith (1886) approach. At a value of h of 0.2m, the SIA (1924) method predicted values of Q_Q only 43% of that suggested by the Kinksvarter and Carter (1957) method.

The considerable differences apparent between the stage-discharge relationships obtained by the three methods highlighted the importance of selecting the most appropriate method for incorporation into PINHEAD. There are pre-established limits to the values of h, h / P, b and P for structures to which they are applied (Table 5.6) and these were used to identify the most suitable method. The SIA method was deemed the most suitable. Possessing the lowest threshold value of h, this method was likely to be the most appropriate for estimating sluice discharge at low hydraulic gradients. By choosing the method with the smallest range of variation in sluice discharge with h, potential inaccuracies associated with applying methods for the estimation of weir discharge to sluices were also addressed. Penning board structures are not characterised by a sharp, thin crest, and an added control on discharge will therefore be the conditions imposed on the flow region by a wooden surface characteristically 0.05 m wide. Further controls on sluice discharge were also expected due to vegetation growth in the channel, a process that can potentially restrict flow by up to 200% (Shaw, 1993).

Author	Sluice Discharge =	*h/P*	*h*	*b*
Hamilton Smith (1886)	$0.581 (1 - 0.1 h / b) b \sqrt{g} h^{3/2}$	<0.5	0.07> <0.6	>0.3
Kindsvater and Carter (1957)	$0.554 (1 - 0.0035 h / p) (b + 0.0025) \sqrt{g} (h+0.001)^{3/2}$	<2	>0.03	>0.15
Swiss Engineers Association (SIA) (1924)	$0.544 [1 + 0.064 (b / B)^2 + 0.00626 - 0.00519 (b / B)^2] / (h + 0.0016) \times [1 + 0.5 (b / B)^4 (h / h + P)^2] b \sqrt{g} h^{3/2}$	<1	0.02> <0.8	-

Table 5.6. Equations for estimating discharge through rectangular weirs with side contractions.

a)

b)

Figure 5.8. (a) Sluice P26 on the SWT Reserve. (b) The form of sluices in other areas of the wetland suggest that methods employed on the SWT Reserve may be transferable.

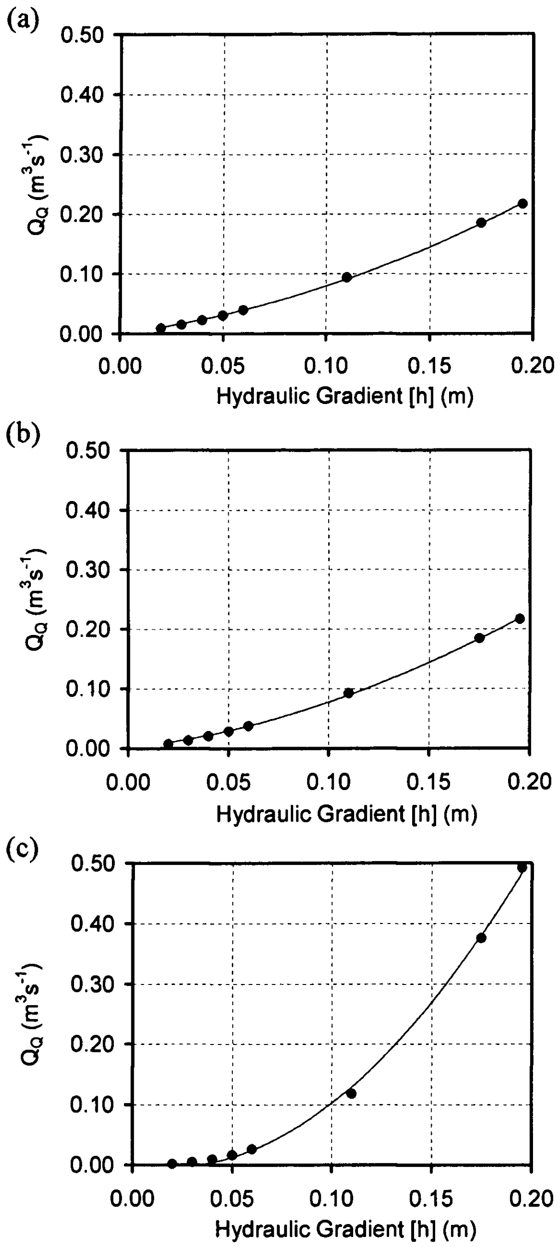


Figure 5.9. Stage-discharge relationships for sluice P26 calculated using (a) Swiss Engineers Association (1924), (b) Hamilton-Smith (1886) and (c) Kinksvarter and Carter (1957) equations (see Table 5.6).

5.3.4.2. Calibrating weir discharge estimates to quantify sluice discharge

To test the validity of the SIA (1924) approach to simulate sluice discharge, the stage-discharge relationship obtained by this method was compared to an alternative estimate of sluice discharge. Water level records for the Field 2 catchment were employed to quantify sluice discharge based on the characteristics of hydrograph recession curves. The method has been termed 'Recession Analysis for Sluice Discharge Estimation' (hereafter termed RASDE). RASDE incorporates the assumption that once recession has commenced, the contributions of runoff to the ditch system have ceased. It is acknowledged that this approach may therefore under-estimate sluice discharge under some conditions. For each hydrograph, the recession is defined as the period between the end of the hydrograph peak and the point of inflexion, as employed in traditional methods of hydrograph separation (Shaw, 1993). The total volume discharged through the sluice (Q_V; in m^3) is then calculated for individual 12-hour periods during each recession to ensure that sufficient data are available to calibrate weir equations. A 12-hour period is chosen because it is the smallest time-step that can be realistically employed using water level charts where 1mm is equivalent to two hours.

In the RASDE, for each 12-hour period, Q_V is calculated by:

$$Q_V = D_{S\ 12\text{-}hr\ Start} - D_{S\ 12\text{-}hr\ End} \qquad \text{(Equation 5.17)}$$

where $D_{S\ 12\text{-}hr\ Start}$ and $D_{S\ 12\text{-}hr\ End}$ are ditch storage at the beginning and end of each 12-hour period. Both values are calculated by application of the level-volume-area relationships for ditches and inundation (Sections 5.2.2 and 5.2.4) to data describing ditch water levels at the start and end of each 12-hour period ($\mathrm{DWL}_{12\text{-hr Start}}$ and $\mathrm{DWL}_{12\text{-hr Start}}$ respectively). The rate of sluice discharge (Q_Q; in m^{3}s^{-1}) is then calculated by:

$$Q_Q = \frac{Q_V}{t} \qquad \text{(Equation 5.18)}$$

where t is 43,200, the number of seconds in 12 hours. Values of Q_Q for each 12-hour period are then plotted relative to the hydraulic gradient (h) to derive a revised stage-discharge relationship, where h is given by

$$h = \mathrm{DWL}_{12\text{-hr Start}} - l_{\mathrm{Sluice}} \qquad \text{(Equation 5.19)}$$

and l_{Sluice} is the sluice level (m OD). For the application of the RASDE on the SWT Reserve, hydrographs were selected from ditch water level records where:

- DWL_{Peak} was greater than sluice level,
- $DWL_{Recess\ End}$ was equivalent, or closely equivalent to the sluice level, and
- there was no rainfall during the recession period.

Based on these criteria, 13 events were identified in water level records. For these events, hydrographs beginning at the start of the recession, and shown relative to a common start water level, are shown in Figure 5.10. Figure 5.10 also shows the hydrographs selected for the analysis of runoff (Section 5.3.5), beginning at the start of the water level rise and peak water levels and shown relative to a common start water level. For each hydrograph, the duration of the antecedent, peak and recession limbs of each hydrograph are summarised in Table 5.7.

The stage-discharge relationship developed by application of the RASDE to the 13 hydrographs is shown in Figure 5.11 relative to that obtained using SIA weir equation. Stage-discharge relationships estimated by the RASDE were characterised by a smaller range of Q_Q with h than the weir equations selected. For the entire range of h, RASDE Q_Q was nearly an order of magnitude smaller than Q_Q estimated by the SIA method (Figure 5.11). Results supported suggestions by Samuels (1993) who has stated that the conveyance of fenland channels is consistently over-estimated. Scatter about the stage-discharge relationship was however considerable (Figure 5.11.b), although segregation of the relationships according to summer (May-September) and winter events (October-April) resulted in a considerable strengthening in the relationship between Q_Q and h (Figure 5.12). The slope of the summer stage-discharge relationship was considerably shallower than that obtained for winter. At a value of h of 0.2m, summer estimates of Q_Q were 46.6% of winter values. Results potentially highlight the important influence of macrophytic vegetation on the conveyance of fenland channels, expressed as seasonal variations in the stage-discharge relationship evident for the sluice (Gilman, 1994).

	Duration (hours)				Ditch Water Level (m OD)			Sluice information	
	TOTAL (t_{TOTAL})	Rise (t_{Rise})	Peak (t_{Peak})	Recession (t_{Recess})	Start of rise ($DWL_{Initial}$)	Peak (DWL_{Peak})	End of Recess ($DWL_{Recess\ End}$)	l_{Sluice} (m OD)	h (m)
07.06.97	72	4	26	42	1.750	1.770	1.750	1.750	0.020
12.06.97	210	38	36	96	1.750	1.830	1.790	1.770	0.060
30.06.97	214	66	48	100	2.020	2.080	2.020	2.020	0.060
06.08.97	140	6	42	92	2.010	2.070	2.010	2.020	0.050
16.08.97	168	42	32	94	2.010	2.060	2.010	2.020	0.040
08.11.97	224	28	36	160	1.980	2.190	2.080	2.020	0.170
24.12.97	136	10	36	90	2.010	2.090	2.030	2.020	0.070
08.01.98	Recess Only	Recess Only	Recess Only	124	Recess Only	2.165	1.990	2.020	0.145
17.01.98	48	6	8	34	2.015	2.040	2.010	2.020	0.020
19.01.98	54	6	8	40	2.005	2.055	2.005	2.020	0.035
23.05.98	322	88	84	150	2.000	2.100	1.990	2.020	0.080
15.06.98	Recess Only	Recess Only	Recess Only	290	Recess Only	2.190	1.990	2.020	0.170
14.11.98	134	8	30	70	1.930	1.970	1.930	1.920	0.050

Table 5.7. Characteristics of hydrographs employed for the development of stage-discharge relationships for sluices on the SWT Reserve.

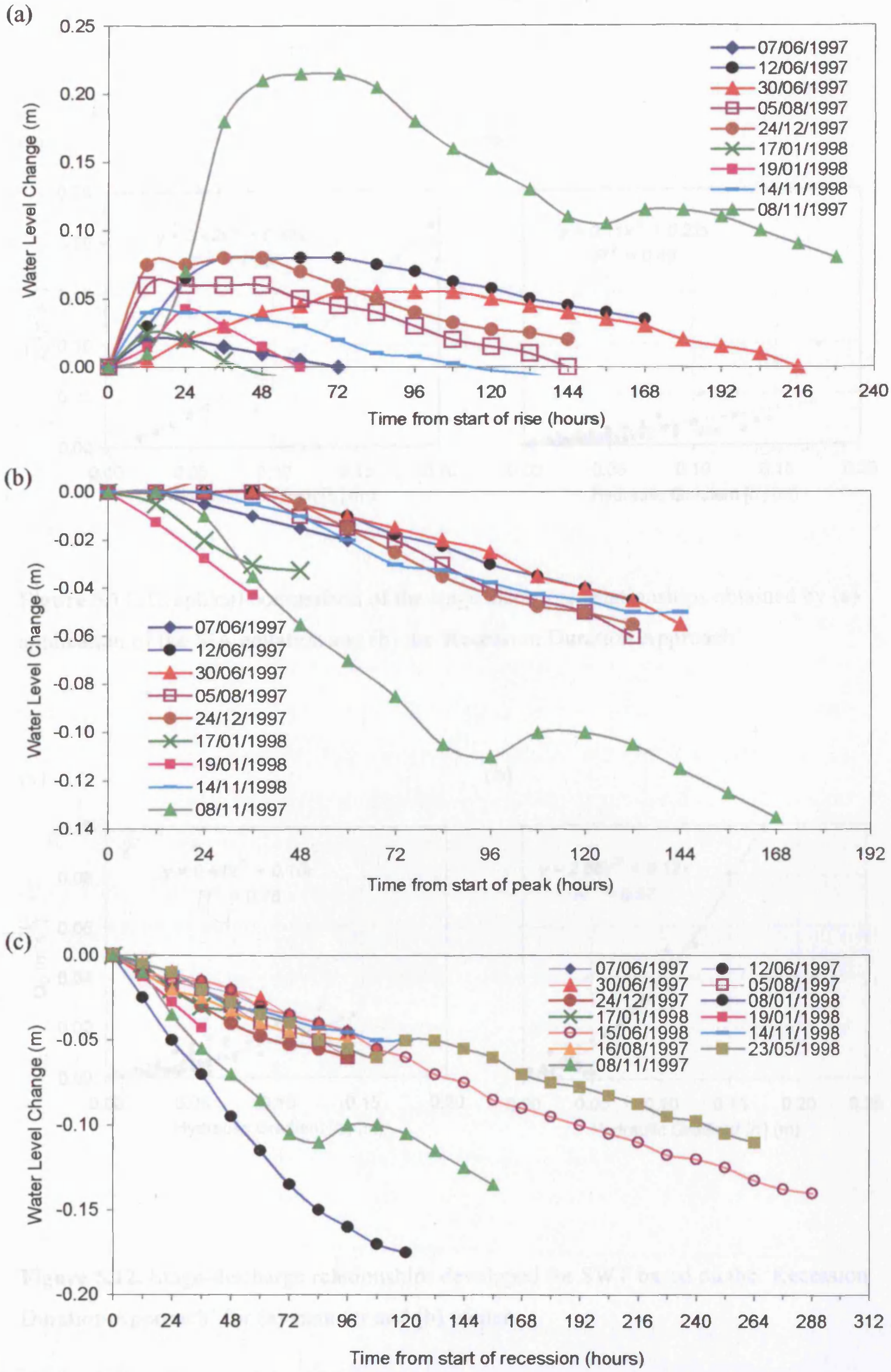


Figure 5.10. Hydrographs from the water level record of the Field 2 catchment on the SWT Reserve employed for the quantification of sluice discharge and runoff. Hydrographs are shown relative to a common start time and water level for (a) initial, (b) peak and (c) recession water levels.

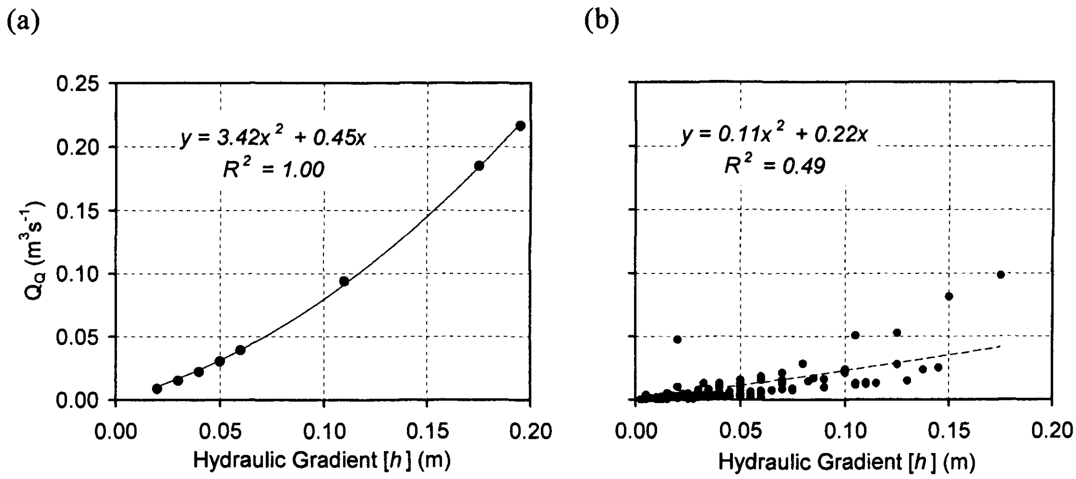


Figure 5.11. Graphical comparison of the stage discharge relationships obtained by (a) application of the SIA equation and (b) the 'Recession Duration Approach'.

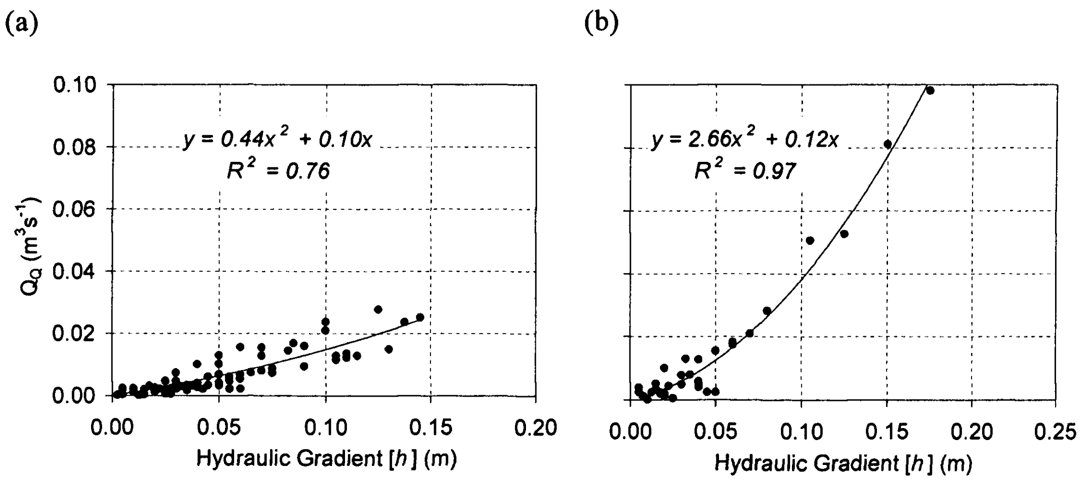


Figure 5.12. Stage-discharge relationships developed for SWT based on the 'Recession Duration Approach' for (a) summer and (b) winter.

5.3.4.3. Simulating sluice discharge in PINHEAD

Findings presented in Section 5.3.4.2 were included as an option in PINHEAD, by incorporating a seasonal approach to the estimation of sluice discharge. In summer:

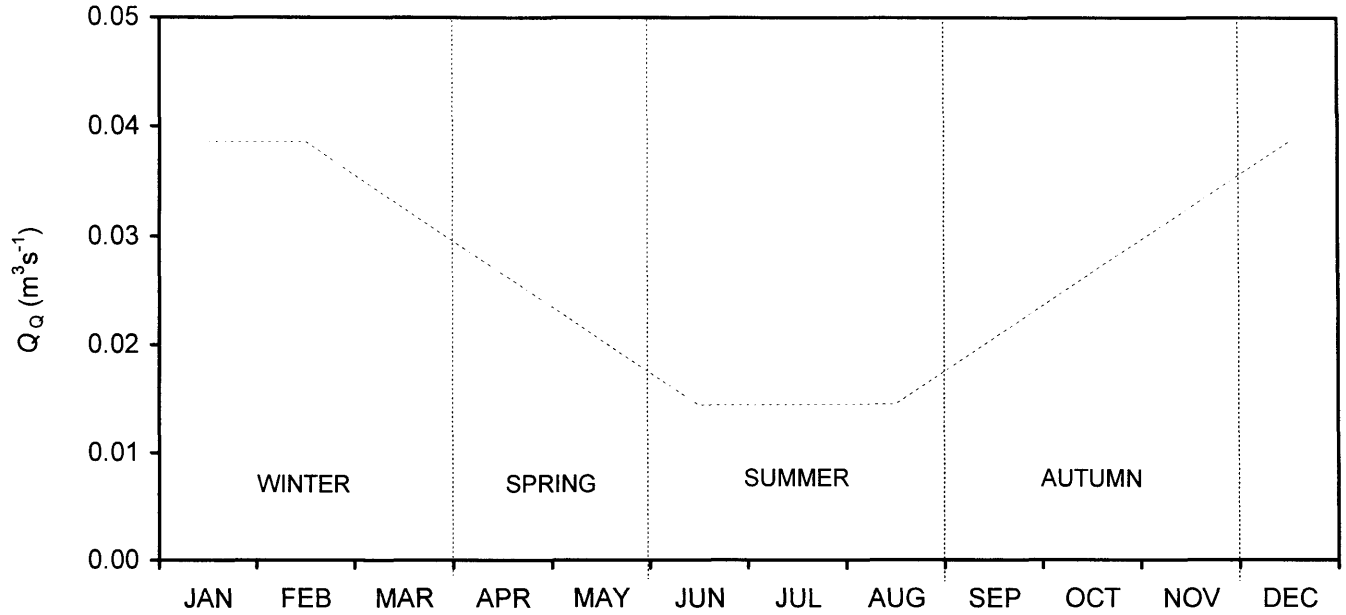
$$Q_Q = 0.44\,h^2 + 0.10\,h \qquad \text{(Equation 5.20)}$$

(Figure 5.12.a) and in winter

$$Q_Q = 2.66\,h^2 + 0.12\,h \qquad \text{(Equation 5.21)}$$

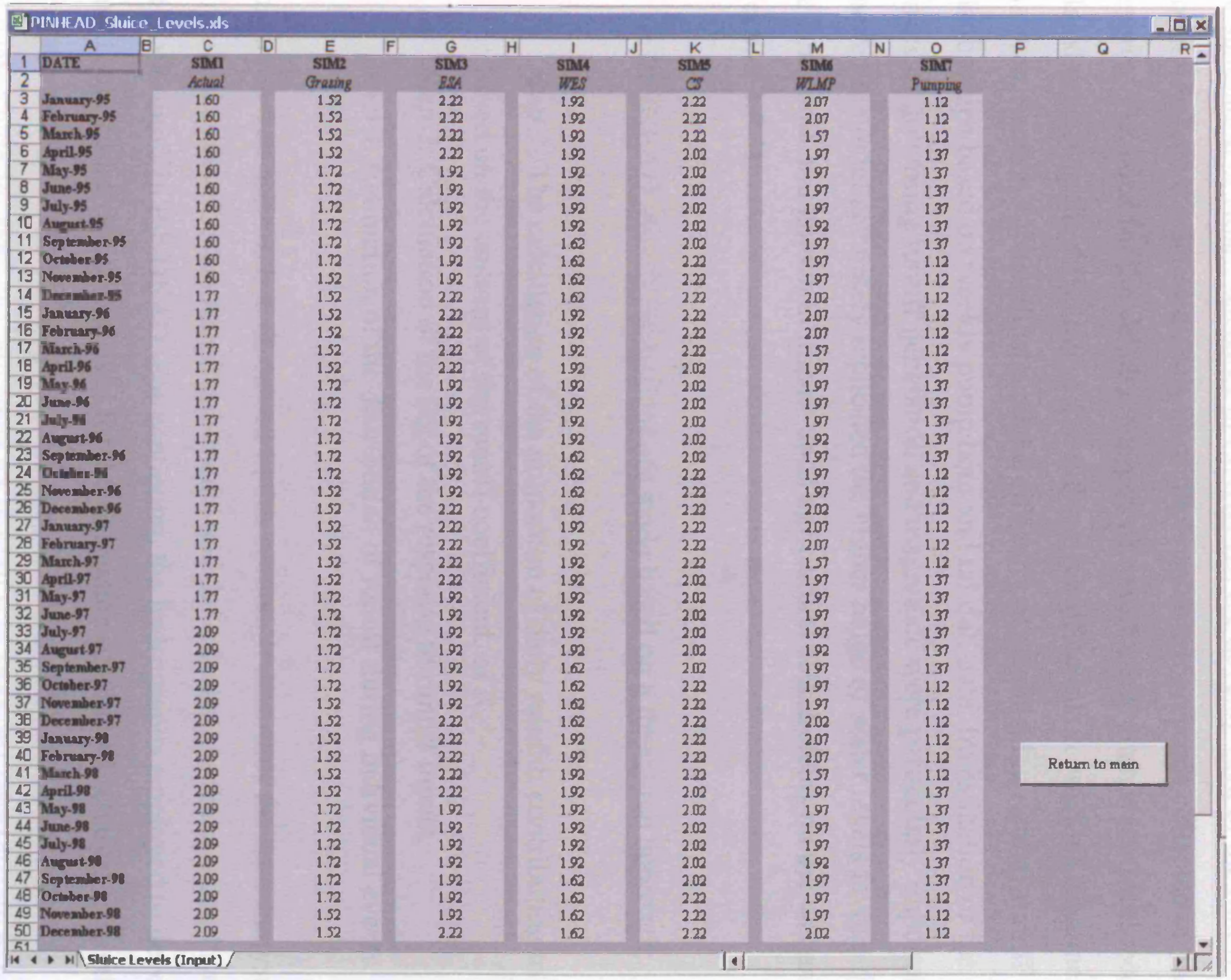
(Figure 5.12.b). A crop growth curve for macrophytic vegetation was employed to determine the specific times of year when either the summer or winter stage-discharge relationships were effective. The winter stage-discharge relationship was applied between December and March. Macrophyte growth usually starts in April (L. Carvalho, Pers. Comm.) and is thickest between June and September (Haslam, 1978). During these months, the summer stage-discharge relationship was applied. Stage-discharge relationships in transitionary periods (April – May and October - November) were interpolated from the summer and winter relationships. In PINHEAD, this is done automatically once values representing the slope of the winter and summer stage-discharge relationships are input to the PINHEAD_Parameters_Input Module (Box 5.7). Cycles of vegetation growth lead to a higher value of Q_Q in the winter than in the summer at equivalent values of h (Figure 5.13). Further details regarding the simulation of vegetation growth and its effect on model performance is given in Section 5.5.1.

A further requirement for the functioning of the sluice discharge sub-model in PINHEAD are time series describing sluice levels during the simulation period. The sluice sub-model operates by relating these values to data describing the sluice elevation (in m OD) on a monthly basis throughout the period modelled. These are input to the PINHEAD_ Sluice_Levels_Input Module, shown in Box 5.8. Based on discussions with the warden of the Reserve, sluice levels between January 1995 and December 1998 could be summarised as 1.60 m OD from January 1995 to December 1995, 1.77 m OD from January 1996 to June 1997, and 2.02 m OD thereafter. For subsequent simulations, monthly sluice level time series for six other management regimes were incorporated to the Module. These alternative sluice management scenarios can be selected from the PINHEAD_Options Module (see Section 6.4.3 and Box 6.2).



AQUATIC PLANT GROWTH PHASE	Dormant	Growth	Maturity	Senescence	Dormant
HYRAULIC RESISTANCE TO FLOW	Low	Increasing	High	Decreasing	Low
SLUICE DISCHARGE =	0.189 *h*	Interpolated	0.089 *h*	Interpolated	0.189 *h*

Figure 5.13. Graphical representation of the method employed to determine the seasonal variations in sluice discharge as a function of vegetation growth in the ditch system. The sluice discharge that would result from a constant hydraulic gradient of 0.10m throughout the year is shown.



PINHEAD_Sluice_Levels.xls

	DATE	SIM1	SIM2	SIM3	SIM4	SIM5	SIM6	SIM7
2		*Actual*	*Grazing*	*ESA*	*WES*	*CS*	*WLMP*	Pumping
3	January-95	1.60	1.52	2.22	1.92	2.22	2.07	1.12
4	February-95	1.60	1.52	2.22	1.92	2.22	2.07	1.12
5	March-95	1.60	1.52	2.22	1.92	2.22	1.57	1.12
6	April-95	1.60	1.52	2.22	1.92	2.02	1.97	1.37
7	May-95	1.60	1.72	1.92	1.92	2.02	1.97	1.37
8	June-95	1.60	1.72	1.92	1.92	2.02	1.97	1.37
9	July-95	1.60	1.72	1.92	1.92	2.02	1.97	1.37
10	August-95	1.60	1.72	1.92	1.92	2.02	1.92	1.37
11	September-95	1.60	1.72	1.92	1.62	2.02	1.97	1.37
12	October-95	1.60	1.72	1.92	1.62	2.22	1.97	1.12
13	November-95	1.60	1.52	1.92	1.62	2.22	1.97	1.12
14	December-95	1.77	1.52	2.22	1.62	2.22	2.02	1.12
15	January-96	1.77	1.52	2.22	1.92	2.22	2.07	1.12
16	February-96	1.77	1.52	2.22	1.92	2.22	2.07	1.12
17	March-96	1.77	1.52	2.22	1.92	2.22	1.57	1.12
18	April-96	1.77	1.52	2.22	1.92	2.02	1.97	1.37
19	May-96	1.77	1.72	1.92	1.92	2.02	1.97	1.37
20	June-96	1.77	1.72	1.92	1.92	2.02	1.97	1.37
21	July-96	1.77	1.72	1.92	1.92	2.02	1.97	1.37
22	August-96	1.77	1.72	1.92	1.92	2.02	1.92	1.37
23	September-96	1.77	1.72	1.92	1.62	2.02	1.97	1.37
24	October-96	1.77	1.72	1.92	1.62	2.22	1.97	1.12
25	November-96	1.77	1.52	1.92	1.62	2.22	1.97	1.12
26	December-96	1.77	1.52	2.22	1.62	2.22	2.02	1.12
27	January-97	1.77	1.52	2.22	1.92	2.22	2.07	1.12
28	February-97	1.77	1.52	2.22	1.92	2.22	2.07	1.12
29	March-97	1.77	1.52	2.22	1.92	2.22	1.57	1.12
30	April-97	1.77	1.52	2.22	1.92	2.02	1.97	1.37
31	May-97	1.77	1.72	1.92	1.92	2.02	1.97	1.37
32	June-97	1.77	1.72	1.92	1.92	2.02	1.97	1.37
33	July-97	2.09	1.72	1.92	1.92	2.02	1.97	1.37
34	August-97	2.09	1.72	1.92	1.92	2.02	1.92	1.37
35	September-97	2.09	1.72	1.92	1.62	2.02	1.97	1.37
36	October-97	2.09	1.72	1.92	1.62	2.22	1.97	1.12
37	November-97	2.09	1.52	1.92	1.62	2.22	1.97	1.12
38	December-97	2.09	1.52	2.22	1.62	2.22	2.02	1.12
39	January-98	2.09	1.52	2.22	1.92	2.22	2.07	1.12
40	February-98	2.09	1.52	2.22	1.92	2.22	2.07	1.12
41	March-98	2.09	1.52	2.22	1.92	2.22	1.57	1.12
42	April-98	2.09	1.52	2.22	1.92	2.02	1.97	1.37
43	May-98	2.09	1.72	1.92	1.92	2.02	1.97	1.37
44	June-98	2.09	1.72	1.92	1.92	2.02	1.97	1.37
45	July-98	2.09	1.72	1.92	1.92	2.02	1.97	1.37
46	August-98	2.09	1.72	1.92	1.92	2.02	1.92	1.37
47	September-98	2.09	1.72	1.92	1.62	2.02	1.97	1.37
48	October-98	2.09	1.72	1.92	1.62	2.22	1.97	1.12
49	November-98	2.09	1.52	1.92	1.62	2.22	1.97	1.12
50	December-98	2.09	1.52	2.22	1.62	2.22	2.02	1.12

Return to main

Sluice Levels (Input)

Box 5.8. The PINHEAD_Sluice_Levels_Input Module.

5.3.5. RUNOFF

5.3.5.1. Estimating runoff magnitude using ditch water level records

For the simulation of runoff within PINHEAD, methods capable of replicating the form and characteristics of hydrographs evident in water level records were required. As in the simulation of sluice discharge in PINHEAD, the development of the runoff sub-model involved the detailed analysis of ditch water level records collected on the SWT Reserve. The need for the detailed analysis of runoff using ditch water level records was related mainly to the difficulty of quantifying runoff based on previous catchment-based analyses. Section 3.4 has illustrated the difficulty of establishing rainfall-runoff relationships based on weekly pump hour and rainfall data. Investigation of the processes governing runoff generation and magnitude were particularly important to ensure the model accurately replicated the higher range of water levels favoured by wetland biota and known to impact upon agricultural stakeholders on the wetland (Section 1.6.3).

In PINHEAD, runoff predictions are made based on a three-step approach:

- *Step 1:* The calculation of the proportion of daily rainfall contributing to runoff based on the concept of the runoff coefficient, or R_c
- *Step 2:* Calculation of the lag of the response to runoff inputs.
- *Step 3:* Estimation of the distribution of runoff during individual events.

This three-step approach is discussed in this section. It describes the data analysis conducted to develop the runoff sub-model and the calculations that are employed to estimate runoff in PINHEAD. For continuity, the hydrographs employed to develop the runoff sub-model were those used for the estimation of sluice discharge. Exceptions were the events on the 16th August 1997 and 23rd May 1998 that were probably associated with lowland feeding since no rainfall was apparent immediately before and throughout their duration. These events were therefore not used in the analysis. Complex storm events with multiple peaks on 8th January 1998 and 15th June 1998 were also discarded. Filtering provided a total of nine hydrographs, shown in Figure 5.14 relative to a common date and start level. Figure 514 is replicated from Figure 5.10.a for clarity.

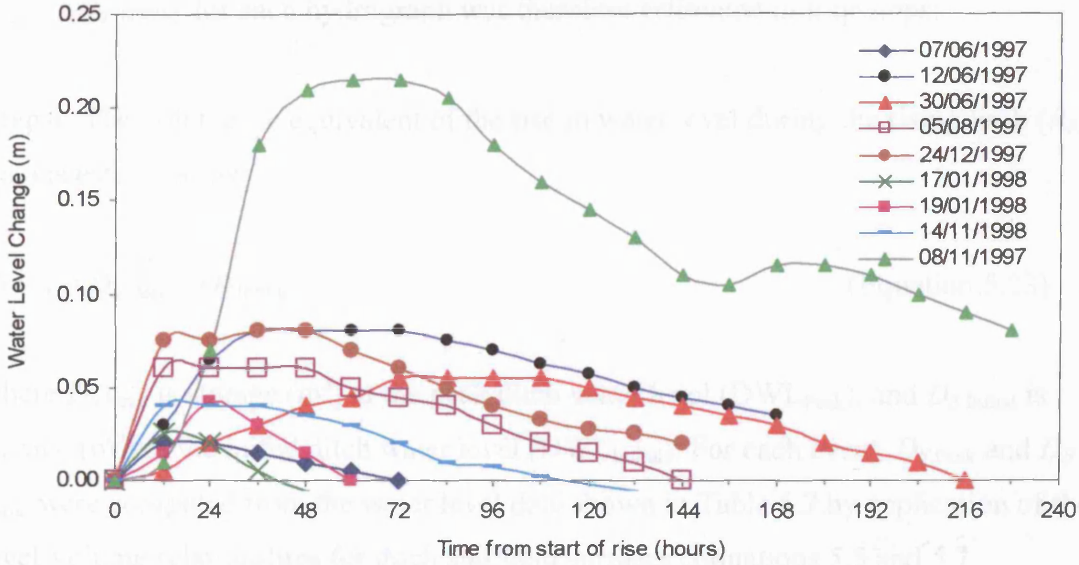


Figure 5.14. Hydrographs employed for the analysis of runoff centred according to the start of water level rise and normalised according to the start water level.

5.3.5.2. Calculating runoff coefficients for the SWT Reserve

For each hydrograph analysed, the runoff coefficient (R_c) was established by

$$R_c = \frac{R_{V\mathrm{TOTAL}}}{C_A\,(P_{\mathrm{Storm}}/1000)} \qquad \text{(Equation 5.22)}$$

where $R_{V\mathrm{TOTAL}}$ is the total runoff generated during the storm, P_{Storm} is the total storm rainfall (mm) taken as Horseye rainfall from the day prior to the hydrograph rise and the end of the hydrograph recession, and C_A is the catchment area. For the Field 2 ditch system, the catchment area is 492,140 m^2 (Figure 5.5). The estimation of $R_{V\mathrm{TOTAL}}$ adopted a similar approach to hydrograph analysis in riverine systems, but accounted for differences in the processes each of the hydrograph components (the rise, peak and recession) represent. For example, an important aspect of hydrographs evident in water level records collected on the SWT Reserve is that during the hydrograph rising limb and peak, runoff generation and sluice discharge occur simultaneously. Consequently, sluice discharge estimates are required for the estimation of $R_{V\mathrm{TOTAL}}$.

$R_{V\,\mathrm{TOTAL}}$ for each hydrograph was therefore estimated in four steps:

Step 1. The volumetric equivalent of the rise in water level during the rising limb (R_{Rise}) (m^3) is estimated by:

$$R_{V\,\mathrm{Rise}} = D_{S\,\mathrm{Peak}} - D_{S\,\mathrm{Initial}} \qquad \text{(Equation 5.23)}$$

where $D_{S\,\mathrm{Peak}}$ is storage (m^3) at the peak ditch water level ($\mathrm{DWL_{Peak}}$), and $D_{S\,\mathrm{Initial}}$ is storage (m^3) at the initial ditch water level ($\mathrm{DWL_{Initial}}$). For each event, $D_{S\,\mathrm{Peak}}$ and $D_{S\,\mathrm{Initial}}$ were computed from the water level data shown in Table 5.7 by application of the level-volume relationships for ditch and field surfaces (Equations 5.5 and 5.7 respectively).

Step 2. Sluice discharge (m^3) during the equivalent period ($Q_{V\,Rise}$) is calculated based on the duration of the rising limb (t_{Rise}) (in seconds), and application of the stage-discharge relationships presented in Section 5.3.4.3. Due to seasonal variations in the stage-discharge relationships, for summer events:

$$Q_{V\,Rise} = 0.44\,h^2 + 0.10\,h\,t_{\mathrm{Rise}} \qquad \text{(Equation 5.24)}$$

and for winter events:

$$Q_{V\,Rise} = 2.66\,h^2 + 0.12\,h\,t_{\mathrm{Rise}} \qquad \text{(Equation 5.25)}$$

h, the elevation above the sluice level or the hydraulic gradient (5.3.4.1), is given by:

$$h = \mathrm{DWL_{Peak}} - \mathrm{DWL_{Sluice}} \qquad \text{(Equation 5.26)}$$

$\mathrm{DWL_{Peak}}$ has been previously defined and $\mathrm{DWL_{Sluice}}$ is the sluice level (in m OD). Values of h, t_{Rise}, $\mathrm{DWL_{Peak}}$ and $\mathrm{DWL_{Initial}}$ for all the events employed for the analysis of runoff on the SWT Reserve are shown in Table 5.7.

Step 3. Runoff during the hydrograph peak ($R_{V\,\mathrm{Peak}}$) is calculated. Because water levels during the hydrograph peak remain constant, runoff generation and sluice discharge can be considered as equivalent, and $R_{V\,\mathrm{Peak}}$ is calculated directly from the stage-discharge relationships developed for sluice discharge. Due to seasonal variations in the stage-discharge relationship therefore, in summer:

$$R_{V\,\mathrm{Peak}} = 0.44\, h^2 + 0.10\, h\, t_{\mathrm{Peak}} \qquad \text{(Equation 5.27)}$$

and in winter:

$$R_{V\,\mathrm{Peak}} = 2.66\, h^2 + 0.12\, h\, t_{Peak} \qquad \text{(Equation 5.28)}$$

where t_{Peak} is the duration of the peak (s), and h is given by Equation 5.26. Values of t_{Peak} for all storms analysed are given in Table 5.7.

Step 4. Assuming that once water levels begin to recede, runoff generation has ceased, the combination of Steps 1 to 3 is employed to estimate $R_{V\,\mathrm{TOTAL}}$ by

$$R_V = R_{V\,\mathrm{Rise}} + Q_{V\mathrm{Rise}} + R_{V\,\mathrm{Peak}} \qquad \text{(Equation 5.29)}$$

For all hydrographs analysed, values of $R_{V\,\mathrm{Rise}}$, $R_{V\,\mathrm{Peak}}$ and $R_{V\,\mathrm{TOTAL}}$ are summarised in Table 5.8. The runoff coefficients obtained from these data by application of Equation 5.22 are also provided.

For the nine events considered, the values of R_c obtained ranged from 3.9% to 62.8%. The maxima, minima, mean and range of coefficients calculated for the SWT Reserve were all approximately equivalent to values observed on Newborough Fen, previously shown in Table 1.10. Further support for the use of ditch water level records to estimate runoff magnitude was provided by the correspondence between the mean value of R_c obtained on the SWT Reserve ($R_{c\,\mathrm{Mean}}$ = 37.4%) and runoff coefficients on the Willingdon Level. The Willingdon Level, a wet grassland site 2 km south west of SWT Reserve, is dominated by similar soil types to those evident on the Pevensey Levels (Section 2.3). Based on the Flood Studies Report approach (NERC, 1975), Binnie and Partners (1988) have identified a mean runoff coefficient of 42% as applicable to that site.

	P_{Storm} (mm)	$C_A\ P_{Storm}$ (m^3)	Response Time (days)	Runoff Duration (days)	$R_{V\,Rise}$ (m^3)	$Q_{V\,Rise}$ (m^3)	$R_{V\,Peak}$ (m^3)	$R_{V\,TOTAL}$ (m^3)	Runoff coefficient R_c (%)
07.06.97	24.9	12254	1 - 2	1 - 2	281.2	26.8	174.0	482.0	3.9
12.06.97	15.3	7530	1 – 2	> 3	1167.6	236.8	224.3	1628.7	21.6
30.06.97	20.6	10138	1 – 2	> 3	2375.2	1567.6	1140.1	5082.9	50.1
06.08.97	47.9	23574	1 – 2	1 - 2	1799.5	105.2	736.5	2641.1	11.2
16.08.97	0.5	246	Feeding	Feeding	1262.2	564.0	429.7	2255.9	Feeding
08.11.97	83.6	39913	<1	> 3	18807.9	2745.9	3530.5	25084.3	62.8
24.12.97	17.7	8317	<1	1 - 2	3473.3	342.7	1233.6	5049.5	60.7
08.01.98	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only
17.01.98	8.7	1772	<1	< 1	522.3	108.9	145.2	776.3	43.8
19.01.98	5.2	2559	1 – 2	< 1	1168.8	175.8	234.4	1578.9	61.7
23.05.98	28.9	14223	Feeding	Feeding	4677.3	2851.2	2721.6	10250.2	Feeding
15.06.98	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only	Recess Only
14.11.98	11.5	5463	1 - 2	1 - 2	698.5	119.7	299.3	1117.6	20.5

Table 5.8. Summary table of the main characteristics of events considered in the analysis of runoff magnitude.

5.3.5.3. Predicting the runoff coefficient in PINHEAD

For implementation within the PINHEAD runoff sub-model, the relationship between R_c and a variety of indices describing antecedent catchment conditions was investigated. To limit model data requirements, the indices employed were all associated with data required by other PINHEAD sub-models. Indices employed described antecedent rainfall, total storm rainfall, soil moisture deficit and antecedent ditch water levels. Results are shown in Figure 5.15. The highest coefficient of determination was provided by the relationship between R_c and 30-day antecedent rainfall (R^2=0.48; Figure 5.15.c). Consequently, this index was chosen for incorporation within the runoff sub-model of PINHEAD. By selecting 30-day antecedent rainfall to provide estimates of Rc within the runoff sub-model, one of the main model objectives of model construction was satisfied. An important advantage of this method above others considered was the spatial and temporal availability of rainfall data on the site, and the fact that rainfall data were required by the model for the functioning of the rainfall sub-model.

The analysis also provided an indication of the likely influence of raising ditch water levels on the processes governing runoff generation. The most likely hypothesis for the positive relationship between R_c and antecedent water levels shown in Figure 5.15.f is that higher ditch water levels promote a more extensive contributing area, probably by saturating the grips. Indeed, the five highest estimates of R_c obtained (all greater than 40%) were all associated with water levels close to, or greater than, 2.00m OD. For hydrographs with initial water levels less than 2.00m, values of R_c obtained were generally below 25%. One outlier in this relationship was the event of 6th August 1997. For this event, antecedent water levels were 2.01m OD but R_c was only 11.6%. According to field notes, during this period two boards were removed from sluice P26. For this event, the method applied for the estimation of V_{TOTAL} was therefore unsuitable, as sluice discharge would have been greater than that predicted based on the characteristic sluice level management regime, leading to an under-estimate of the runoff coefficient. Indeed, removal of the outlier resulted in a considerably higher coefficient of determination for the relationship (R^2= 0.74). As a result, an option to use antecedent ditch water levels to estimate runoff contributions was incorporated in the Options Module (Box 5.9), although it was not used in subsequent simulations. The relationship was thought unlikely to be effective where water levels are actively managed for agriculture, limiting the scenario-testing capabilities of the model.

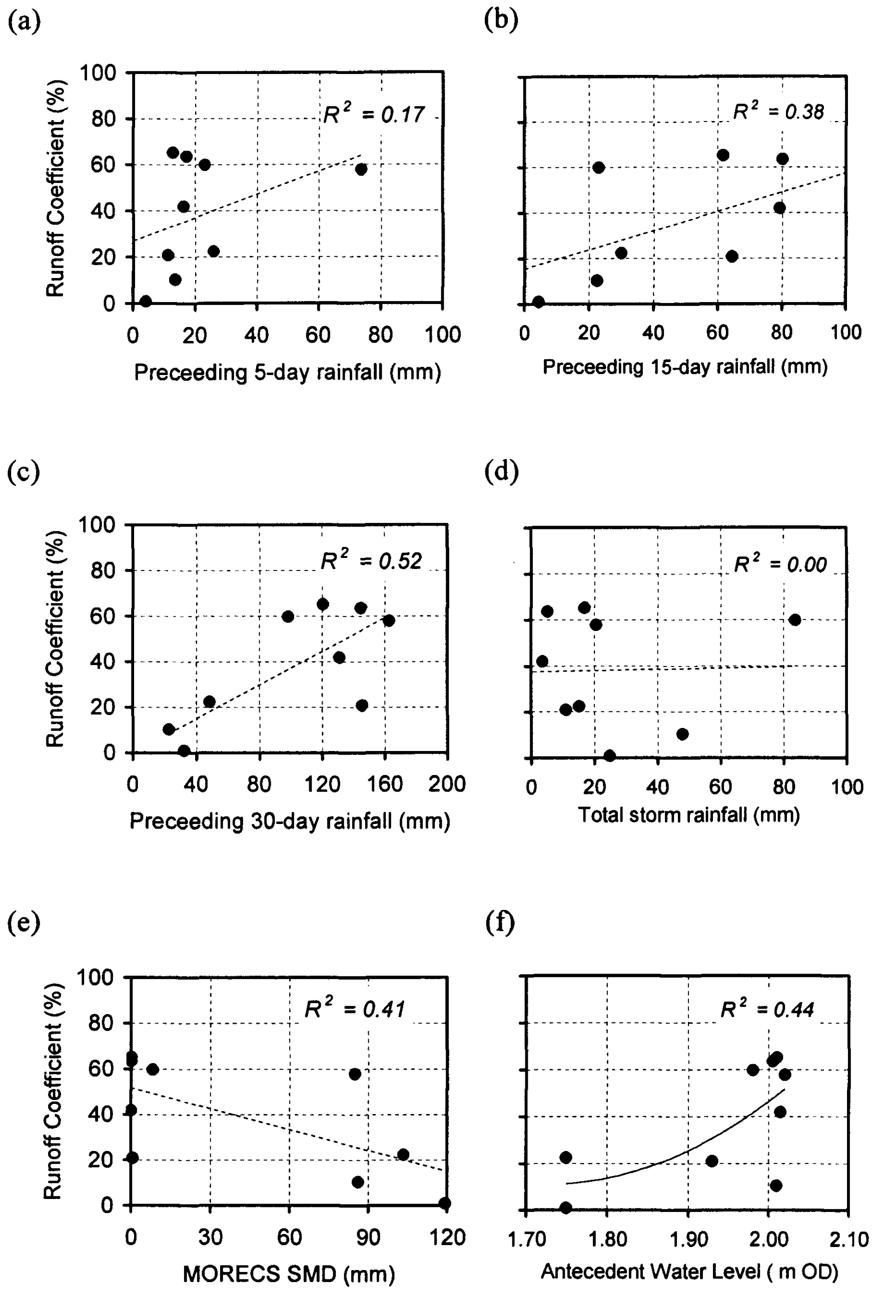


Figure 5.15. The relationship between the runoff coefficient (R_c) obtained by the recession duration method and hydrological indices describing antecedent catchment conditions.

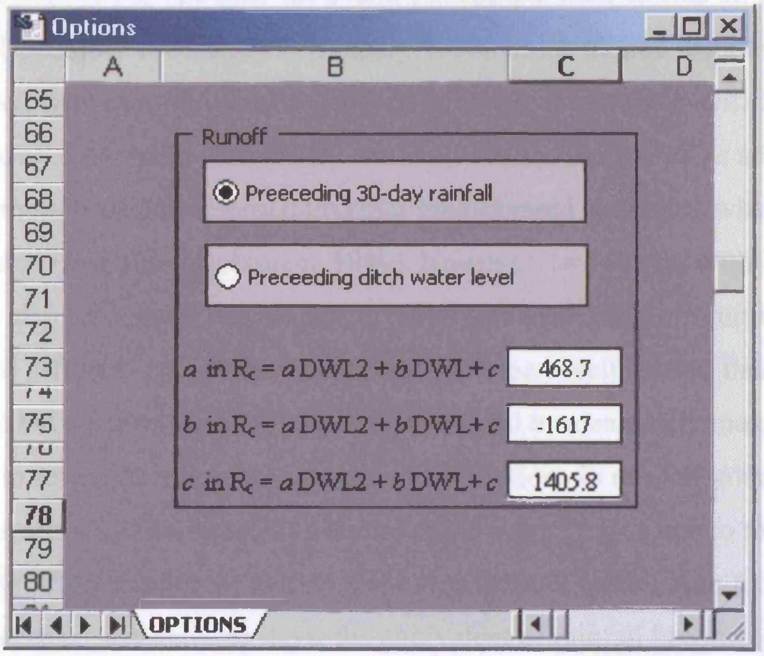
The specific method employed to calculate R_c within the model can be selected in the 'Runoff' frame of the PINHEAD_Options Module of the main model screen (Box 5.9). For the use of the relationship with 30-day preceding rainfall, the slope of the relationship needs to be provided in the PINHEAD_Parameters_Input Module (Box 5.7). On the SWT Reserve:

$$R_c = 0.37\,P_{30\text{-Day}} \qquad \text{(Equation 5.30)}$$

where $P_{30\text{-Day}}$ is the rainfall in the 30-day antecedent rainfall (mm). For the use of initial ditch water levels to estimate the value of R_c, values of *a*, *b* and *c* in second-degree polynomial regression equation taking the form of Equation 5.2 relating antecedent ditch water levels and R_c is input to cells specified in the Runoff frame of the PINHEAD_Options Module (Box 5.9). On the SWT Reserve, and following the removal of the main outlier for the reasons discussed,

$$R_c = 468.7\,DWL_{\text{Initial}}^{2} - 1617.0\,DWL_{\text{Initial}} + 1405.7 \qquad \text{(Equation 5.31)}$$

For all the hydrographs considered, the value of DWL_{Initial} is given in Table 5.7.



Box 5.9. The runoff frame in the PINHEAD_Options Module.

5.3.5.4. Response times and runoff distribution during individual rainfall events

Another aspect that required consideration during the development of the runoff sub-model in PINHEAD was the response time of the ditch system to rainfall inputs (Step 2 in Section 5.3.5.1) and the estimation of the distribution of runoff during individual storms (Step 3 in Section 5.3.5.1). In the context of PINHEAD, the term response time is used to define the period between the depth centroid of rainfall and the initiation of water level rise. As for runoff coefficients, predictive methods for the estimation of response times and runoff distribution were established based on the analysis of ditch water level records collected on the SWT Reserve. Indices for their estimation were established from the hydrographs employed for the calculation of runoff magnitude. A particular focus was on the analysis of the duration of each of the hydrograph components to the nearest day, since as previously specified the model operates on a daily time-step. As with other methods incorporated within the model, the development of predictive methods for the estimation of response time and runoff distribution was based on data required by other PINHEAD sub-models.

For each of the storms considered in the analysis, response times have been summarised in Table 5.8. For most storms employed in the analysis (six of the nine storms), the response time was between one and two days, the equivalent of rainfall becoming effective the day after the event. Exceptions were storms on the 8th November 1997, 24th December 1997 and 17th January 1998, which all had response times of less than one day, with rainfall becoming effective on the day of the event. The latter three events were all associated with SMDs less than 10mm, and therefore adhere to traditional models of runoff generation that predict rapid responses when soil storage is fully saturated (Ward and Robinson, 1989). However, two storms with longer response times were also associated with values of SMD less than 10mm, complicating the use of this index to differentiate between the two storm types. Neither was there a clear relationship between response time and total rainfall or mean daily maximum rainfall. Variations in response time were attributed to variations in rainfall intensity during individual storms, although this hypothesis could not be tested due to the discontinuous nature of the AWS rainfall record. As a result, a static response time for all runoff events of between one and two days, the equivalent of rainfall becoming effective the day after the event, was implemented within the runoff sub-model.

For implementation of step 3 in the runoff sub-model (Section 5.3.5.1), the distribution of runoff through time within individual events was examined using the relationship between the cumulative proportion of total storm runoff and time. The relationship between total hydrograph runoff and total hydrograph duration (Figure 5.16.a) did not however allow the classification of hydrograph events. The most appropriate and simple means of classification was based on total runoff duration. As in previous analyses, runoff duration was taken as the combined duration of the rising and peak limbs of individual hydrographs, assuming that once the hydrograph recession had commenced runoff had ceased. The events considered had total durations of either less than one day, between one and two days, or more than three days (Figure 5.16.b). For storms with a duration of more than three days, the first day accounted for between 18% and 50% of total runoff (mean = 31%), the second day for between 29% and 42% (mean = 35%) and the third day for 21% - 40% (mean = 35%). For storms with a runoff duration of one to two days, the first day represented 52% and 80% of total hydrograph runoff (mean = 65%), and the second day between 21% and 40% (mean = 35%).

However, for the implementation of the runoff sub-model, only a distinction between storm runoff duration of less than one day and between one and two days was necessary. This was because all storms with a runoff duration in excess of three days seemed associated with episodes of continuous rainfall, (*eg.* 8th November 1997), or occurred at the beginning of summer and may have been coupled with lowland feeding. In the case of events on 12th June 1997 and 30th June 1997, further support for this hypothesis was provided by the long response times of hydrographs to incoming rainfall, and the large rises in water levels relative to rainfall inputs. Winter events with a duration in excess of three days could be partially considered, in conceptual terms at least, as coalescent hydrographs of storms with a shorter runoff duration. Storms with a runoff duration of less than one day, such as those on the 17th and 19th January 1998, were characterised by the lowest values of P_{Storm} of all hydrographs analysed (8.7mm and 5.2mm respectively), providing an indication of the influence of total storm runoff on hydrograph duration. Within the runoff sub-model, these results were adopted by assuming that when daily rainfall was less than 10mm, runoff duration was one day. When rainfall exceeded this value, 65% of runoff was distributed on the first day and 35% on the second day, the mean values associated with distributions for storms with a runoff duration of between one and two days.

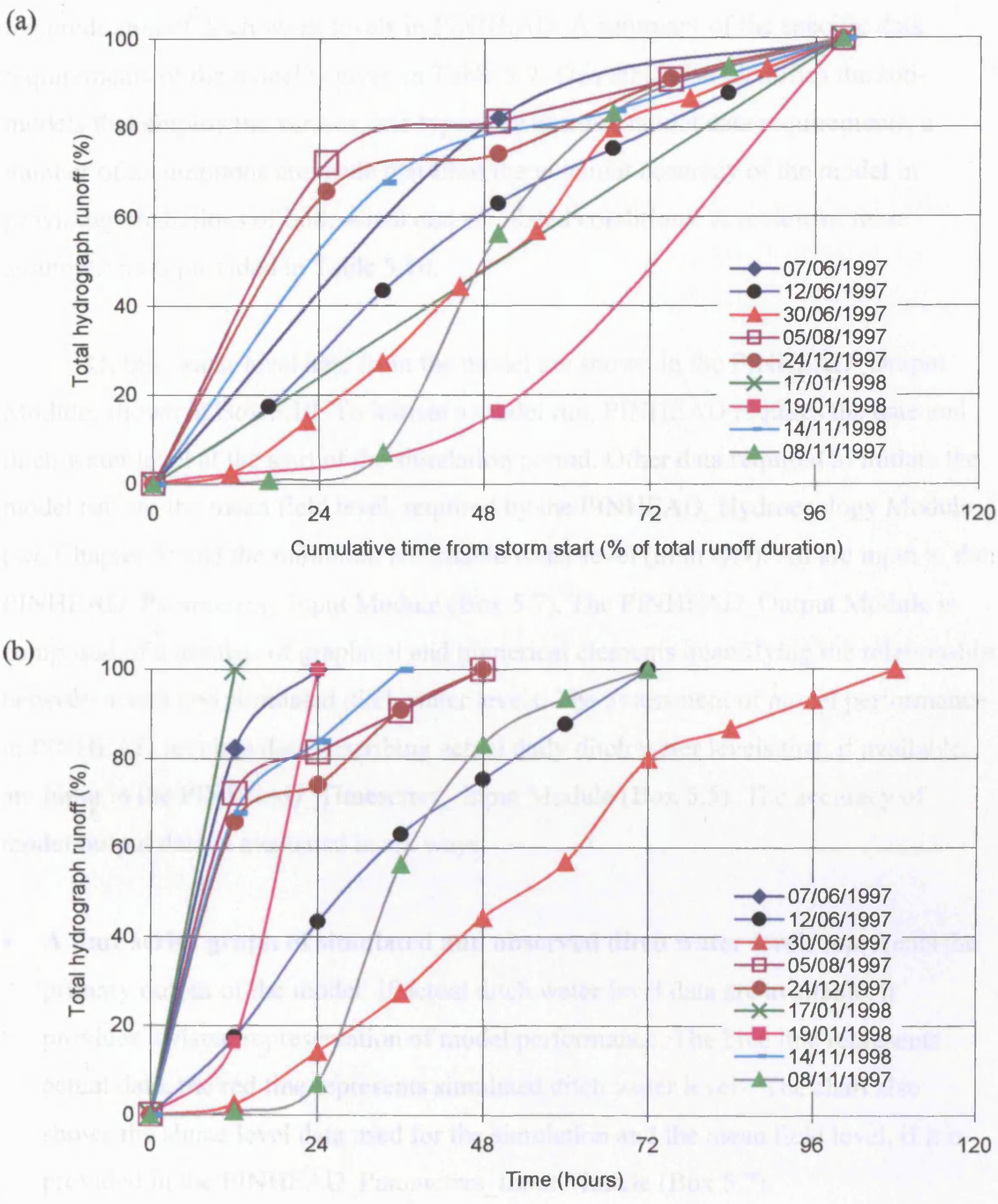


Figure 5.16. The temporal distribution of runoff during individual hydrograph events relative to two indices of runoff duration. The period of runoff duration is defined as that occurring between the beginning of the antecedent limb and the end of the peak limb of each hydrograph.

5.4. Reporting and manipulating results in PINHEAD

Implementation of the procedures described in Sections 5.2 and 5.3 provide the basis for the prediction of ditch water levels in PINHEAD. A summary of the specific data requirements of the model is given in Table 5.9. This table also identifies the sub-models that employ the various data types. By limiting model data requirements, a number of assumptions are made that limit the potential accuracy of the model in providing predictions of both actual and simulated conditions. A review of these assumptions is provided in Table 5.10.

Output water level data from the model are shown in the PINHEAD_Output Module, shown in Box 5.10. To initiate a model run, PINHEAD requires the date and ditch water level at the start of the simulation period. Other data required to initiate the model run are the mean field level, required by the PINHEAD_Hydroecology Module (see Chapter 6) and the minimum recordable water level (in m OD). All are input to the PINHEAD_Parameters_ Input Module (Box 5.7). The PINHEAD_Output Module is composed of a number of graphical and numerical elements quantifying the relationship between actual and simulated ditch water levels. The assessment of model performance in PINHEAD requires data describing actual daily ditch water levels that, if available, are input in the PINHEAD_Timeseries_ Input Module (Box 5.5). The accuracy of model output data is evaluated in six ways:

- **A time series graph of simulated and observed ditch water levels** represents the primary output of the model. If actual ditch water level data are available, it provides a visual representation of model performance. The blue line represents actual data, the red line represents simulated ditch water levels. The chart also shows the sluice level data used for the simulation and the mean field level, if it is provided in the PINHEAD_Parameters_Input Module (Box 5.7).

- **An X-Y plot of simulated and observed ditch water levels** provides the operator with a visual representation of the relationship between simulated results and actual conditions about a 1:1 line representing a perfect model fit. On-screen values of the coefficient of determination (R^2) and the correlation coefficient (where 1.00 is equivalent to the 1:1 line) are also provided as part of the graphical output.

- **The level-frequency plot of simulated and observed ditch water levels** employed in PINHEAD is the equivalent of a flow-frequency distribution curve commonly used in riverine hydrological studies (Shaw, 1993). These data can also be used to quantify the impacts of different water level management scenarios on wetland stakeholders, an approach discussed in Chapter 6.

- **An X-Y plot of the model residuals against observed ditch water levels and a line chart describing the cumulative change in observed and simulated water levels** provide an indication of model bias, allowing visualisation of a specific range of water levels where the predicted levels deviate from observed levels. These charts are accessed by pressing the button labelled 'More Statistics' in the PINHEAD_ Output Module (Box 5.10).

- **Values of F1, F2 and F4** are parameters used for the assessment of model accuracy (Kirkby *et al.*, 1992) and are accessed by pressing the button labelled 'More Statistics' in the PINHEAD_Output Module (Box 5.10). F1 is the root mean square (RMS) error, given by:

  $\Sigma\ (DWL_{Actual} - DWL_{Simulated})^2\ /\ n$ (Equation 5.32)

  F2 is the mean of absolute errors, given by:

  $[\Sigma\ (DWL_{Actual} - DWL_{Simulated})\ /\ n]$ (Equation 5.33)

  F4, the mean difference between observed and simulated water levels is given by:

  $[\Sigma\ (DWL_{Actual}\ /n)] - [\Sigma\ (DWL_{Simulated}\ /\ n)]$ (Equation 5.34)

- **Water level statistics for the entire simulation period.** The maximum, minimum, mean and the range of water levels during the simulation period, and the frequency of occurrence of water levels greater than 2.00 m OD (the inundation threshold water level) and less than 1.40 m OD (the 'dry' level), are automatically calculated by the PINHEAD_Output Module. These data are accessed by pressing the button labelled 'More Statistics' in the PINHEAD_Output Module (Box 5.10). Similar links enable viewing of the Level_Volume and other PINHEAD Modules (Box 5.10).

Data	Source	Sub-model	Input Module
Daily Rainfall	Environment Agency Met Office	Rainfall Runoff	PINHEAD_Timeseries_Input_Data
Daily Evaporation	Tank or Penman Met Office	Evaporation	PINHEAD_Timeseries_Input_Data
Soil Moisture Deficit	Met Office	Ground-surface water interactions	PINHEAD_Timeseries_Input_Data
Hydraulic Conductivity	Field measurement Soil Survey of England Literature (Table 5.15)	Ground-surface water interactions	PINHEAD_Parameters_Input
Sluice Levels	Internal Drainage Board Landowners/managers	Sluice Discharge	PINHEAD_Sluice_Levels_Input
Ditch Cross-sections	Field measurements Newbold *et al* (1989)	Ground surface water interactions. Indirectly related to all sub-models (level-volume-area relationships)	PINHEAD_Level_Volume_Input
Ditch Length	Field measurements Maps	Ground surface water interactions. Indirectly related to all sub-models (level-volume-area relationships)	PINHEAD_Level_Volume_Input
Catchment Area	Field measurements Maps	Runoff	PINHEAD_Parameters_Input
Ditch Water Levels	Internal Drainage Board Monitoring	None, but required for model calibration and validation	PINHEAD_Timeseries_Input_Data
Digital Elevation Model	Field survey	None, but required for inundation level-volume-area relationships	PINHEAD_Inundation_Input

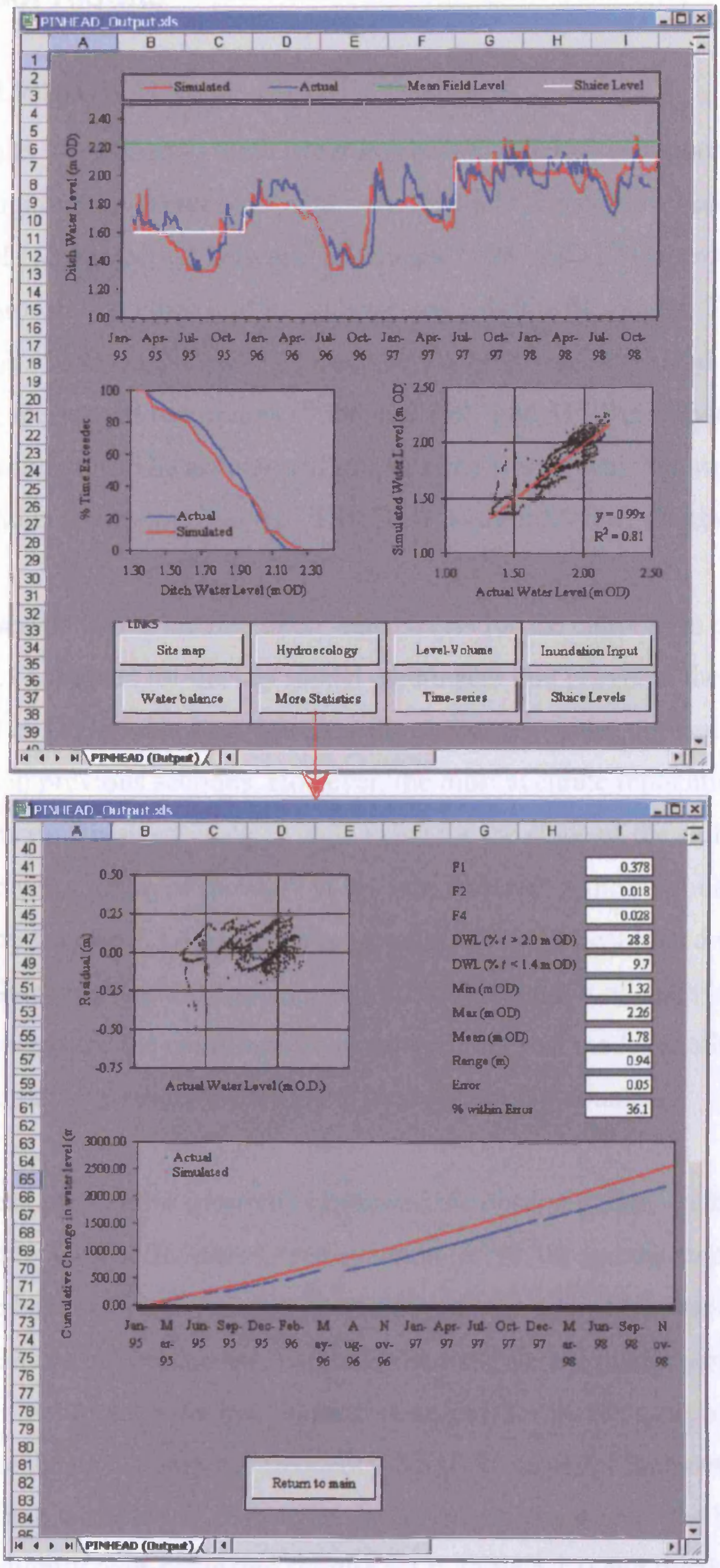
Table 5.9. The data requirements of PINHEAD. Reliant sub-models are specified as are the Modules where these data are input to.

Rainfall sub-model
• Rainfall data describe rainfall over the entire catchment • All rainfall falling within the cross section of a ditch contributes to ditch storage, irrespective of ditch water level
Evaporation sub-model
• Tank evaporation data, adjusted using a time invariant coefficient, are suitable for estimating water loss from the ditch water surface • Tank evaporation data, adjusted using a time invariant coefficient, are suitable for estimating water loss from an inundated field surface
Runoff sub-model
• The rainfall-runoff relationship is dependant on the rainfall on the preceding 30 days • For all storms, runoff becomes effective the day after the rainfall event • In summer, runoff duration is 1 day if daily rainfall is less than 20 mm, otherwise 2 days • In winter, runoff duration is 1 day if daily rainfall is less than 10mm, otherwise 2 days • If runoff duration is 2 days, 65% of total runoff is conveyed to the ditch in day $t1$ and 35% on day $t2$
Ground-Surface Water Interactions sub-model
• Hydraulic conductivity is uniform across the catchment • The area over which the interactions between groundwater and surface water occur is proportional to the hydraulic gradient • The hydraulic gradient is proportional to the Soil Moisture Deficit
Sluice Discharge sub-model
• One sluice controls discharge from the lowland ditch system • The duration of hydrograph recessions can be employed to estimate sluice discharge
Level-volume-area relationships
• There are no more than two distinct ditch types within the ditch catchment • Ditch network sub-catchments in wet grassland areas can be delineated using sluices, blocked ends, roads and different order ditches as boundaries

Table 5.10. Assumptions incorporated within the PINHEAD model.

a)

b)



Box 5.10. Components of the PINHEAD_Output Module in PINHEAD. Only (a) is visible from the main model screen, that also includes links to remaining model Modules. (b) is accessed by pressing the button labelled 'More statistics' in (a).

5.5. Model results

5.5.1. CALIBRATION

Model results were fitted to ditch water levels in the Field 2 catchment of the SWT Reserve (Figure 5.5). Water level data from the Field 2 water level recorder wasavailable for the period between 1st January 1995 and 31st December 1998. The split sample approach was employed to calibrate and validate the model. The first two years of the period (1st January 1995- 31st December 1996) were employed for model calibration, the second two years (1st January 1997 and 31st December 1998) were used for model validation. The accuracy of output ditch water level data was assessed based on the methods incorporated in the PINHEAD_Output Module (Section 5.4).

Observed and simulated ditch water levels for the calibration period are shown in Figure 5.17. Values for the key model parameters that provided the best model fit are reproduced in Table 5.11. In most cases, the parameter values applied were those established in previous sections. However, the most accurate replication of observed water levels was obtained when a single value for the slope of the sluice-discharge relationship was employed throughout the year (instead of the seasonal approach supported by Section 5.3.4.3), and when a value of 90% of estimated sluice discharge was used. Since the method implemented for the estimation of runoff (Section 5.3.5) already accounts for the contributions of rainfall, the best model results were obtained with the rainfall sub-model switched off to avoid double-counting.

Model predictions generally replicated the observed daily water levels closely, as indicated by the coefficient of determination (R^2=0.78) and the slope of the relationship between observed and simulated water levels, which was close to unity. Table 5.12 shows calibration and validation statistic for the model period. Values of F1, F2 and F4 in particular were low. Particular support for the accuracy of output data was the correspondence between the frequency-duration curves of both observed and simulated ditch water levels. However, the frequency-duration of the highest water levels was slightly over-estimated (Figure 5.17; Table 5.12) and the frequency-duration of the lowest water levels was slightly under-estimated (Table 5.12), although overall these differences were small supporting the validity of the model.

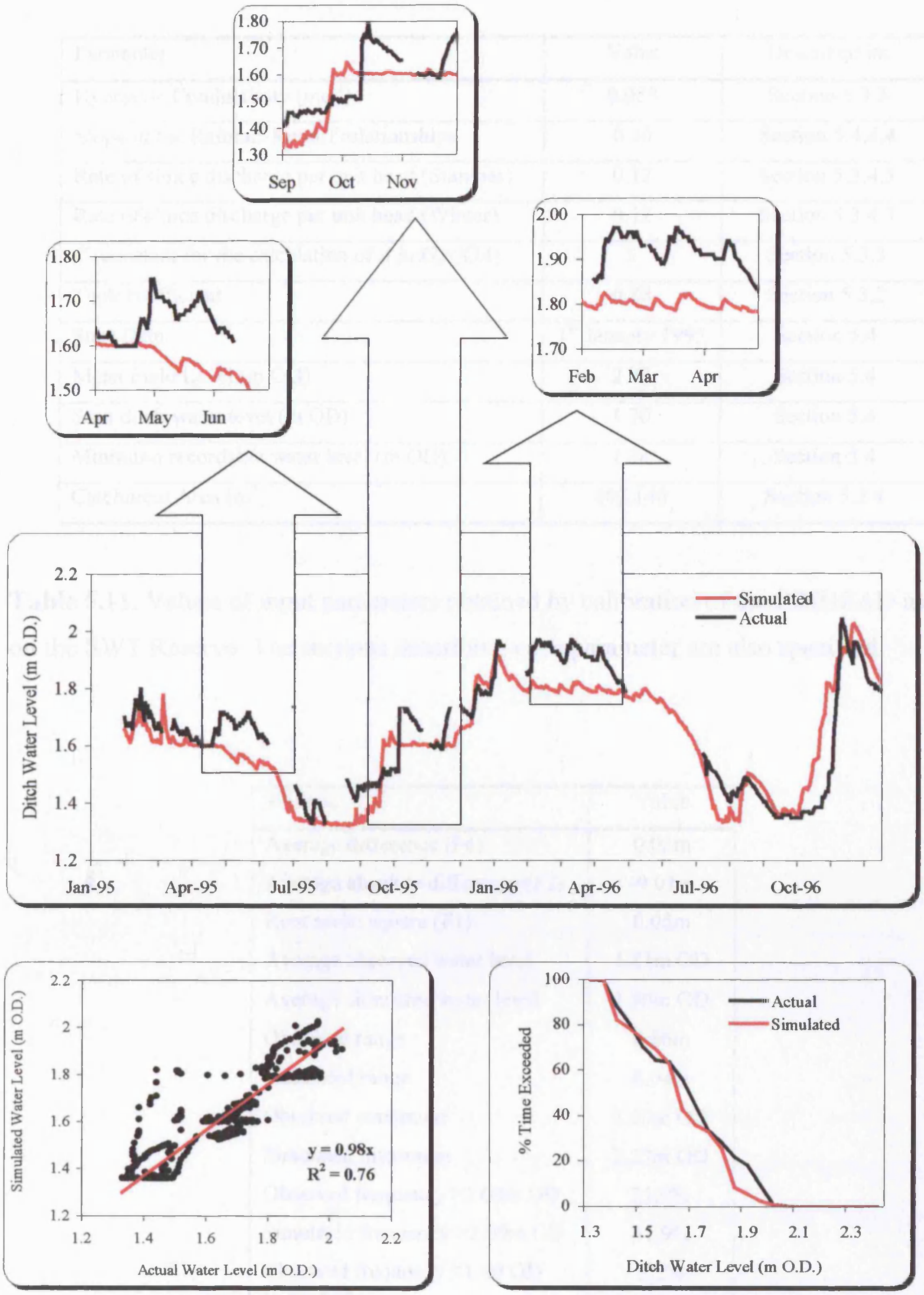


Figure 5.17. Actual and predicted water levels on the SWT Reserve for the calibration period (1st January 1995 to 31st December 1996), highlighting the main inaccuracies associated with model predictions.

Parameter	Value	Described in:
Hydraulic Conductivity (md^{-1})	0.058	Section 5.3.3
Slope of the Rainfall-Runoff relationships	0.36	Section 5.4.4.4
Rate of sluice discharge per unit head (Summer)	0.12	Section 5.3.4.3
Rate of sluice discharge per unit head (Winter)	0.12	Section 5.3.4.3
C (constant for the calculation of A in G_v=KIA)	3	Section 5.3.3
Tank coefficient	0.88	Section 5.3.2
Start Date	1st January 1995	Section 5.4
Mean Field Level (m OD)	2.22	Section 5.4
Start ditch water level (m OD)	1.70	Section 5.4
Minimum recordable water level (m OD)	1.36	Section 5.4
Catchment Area (m^2)	492,140	Section 5.2.4

Table 5.11. Values of input parameters obtained by calibration of the PINHEAD model on the SWT Reserve. The sections describing each parameter are also specified.

Statistic	Value
Average difference (F4)	0.02m
Average absolute difference (F2)	-0.01m
Root mean square (F1)	0.05m
Average observed water level	1.81m OD
Average simulated water level	1.80m OD
Observed range	0.86m
Simulated range	0.94m
Observed maximum	2.20m OD
Simulated maximum	2.27m OD
Observed frequency >2.00m OD	21.9%
Simulated frequency >2.00m OD	31.9%
Observed frequency <1.40 OD	5.2%
Simulated frequency <1.40 OD	10.4%

Table 5.12. Statistics described in Section 5.4 applied to the relationship between simulated and actual water levels 1995-1998 for the Field 2 catchment of the SWT Reserve.

The main model inaccuracies regarded individual hydrograph events. For example, water level peaks on the 30th April 1995 and 25th May 1995 were poorly replicated (Figure 5.17). For both events, the limited rainfall apparent before and during the events suggested increases in water levels may be at least partly associated with lowland feeding, a process not currently incorporated in the model. For other events, the calibrated model did not replicate the timing of hydrograph peaks appropriately. For an event on the 15th October 1995 the simulated peak occurred before the predicted peak (Figure 5.17). This probably relates to the static value of one day incorporated in the model to estimate the response time of ditches to runoff events (Section 5.3.5.4).

5.5.2. VALIDATION

Changes to water level management practices on the site between the calibration and validation periods presented some problems with regards to the accuracy of model output data. Sluice P26 was re-profiled in June 1997, increasing the maximum attainable water levels on the Reserve by 0.4 m. As a result, water levels throughout the validation period were considerably higher than during the calibration period, leading to difficulties in validating the model for the replication of the highest water levels. As a result, some hydrographs during the validation period were poorly replicated, most notably the peaks on 15th February 1997, 28th May 1997 and 15th June 1997 (Figure 5.18). However, both the magnitude of the peaks, and in the case of the latter two events, the timing of the increases in water levels, suggested that these events may have been associated with feeding, a potential influence on the hydrology of the SWT Reserve that has been previously noted for the calibration period (Section 5.5.1).

A more common inaccuracy during the validation period was the inadequate replication of the recession limbs of individual hydrographs. This tended to occur when peak water levels were in excess of the threshold inundation level (2.00 m OD), water levels that were relatively infrequent during the calibration period. This was evident for both individual hydrographs, such as troughs in October 1997 and March 1998, and for longer periods. During most of the summer in 1998 actual water levels receded at a greater rate than those predicted by the model (Figure 5.18). For summer events, differences between observed and simulated water levels could be potentially ascribed to the use of tank evaporation data to simulate evaporative losses during inundated conditions.

Although evaporation tanks are thought to over-estimate the actual evaporative loss due to their small surface area and advection through their side walls (Section 4.1), results presented in Chapter 4 suggest that the use of tank evaporation data to simulate evaporative loss from an inundated wetland surface on the Pevensey Levels may actually lead to an under-estimate of evaporative loss. This is because results provided by the Hydra have indicated that, at levels in excess of 1.85m OD, the actual evaporative loss exceeds the reference potential evaporation rate given by the Horseye evaporation tank (Section 4.7.4 and Figure 4.9.f). Potential support for the validity of these findings was provided by the closer replication of simulated water level recessions for individual events, as well as for longer periods such as the summer of 1998, when evaporation rates were adjusted based on the methodology described in Section 4.7.4 (Figure 5.18). As previously stated this methodology has been included as an option within the PINHEAD_Options module (Section 5.3.2). Implementation of this method also provided a closer replication of the actual frequency-duration curve and a strengthening of the relationship between actual and simulated water levels during the validation period (Figure 5.18).

For other events, where recessions were inappropriately replicated, this could potentially be ascribed to the responses of local landowners to climatic conditions. This is likely to be a practice associated mainly with the winter months, when local farmers will seek to limit inundation of field surfaces that may result in disease in de-pastured stock, and also spring waterlogging that can have an influence on pasture productivity. On the SWT Reserve, this can be illustrated by considering the recession commencing on 19th January 1998 (Figure 5.18). This recession describes water level change following a period of high water levels. For the three weeks preceding the beginning of the recession water levels exceeded the inundation threshold level on 18 days, reaching a maximum water level of 2.17m OD on the 5th January 1998. Field notes taken during a visit to the site on the 21st January 1998, indicate that during the recession period, a number of boards had been removed from sluice P26 (Section 5.3.4.1). This highlights the importance of landowner involvement in the provision of the sluice management data required by PINHEAD and identifies the difficulties associated with the accurate estimation of water levels where detailed data describing sluice management are not available.

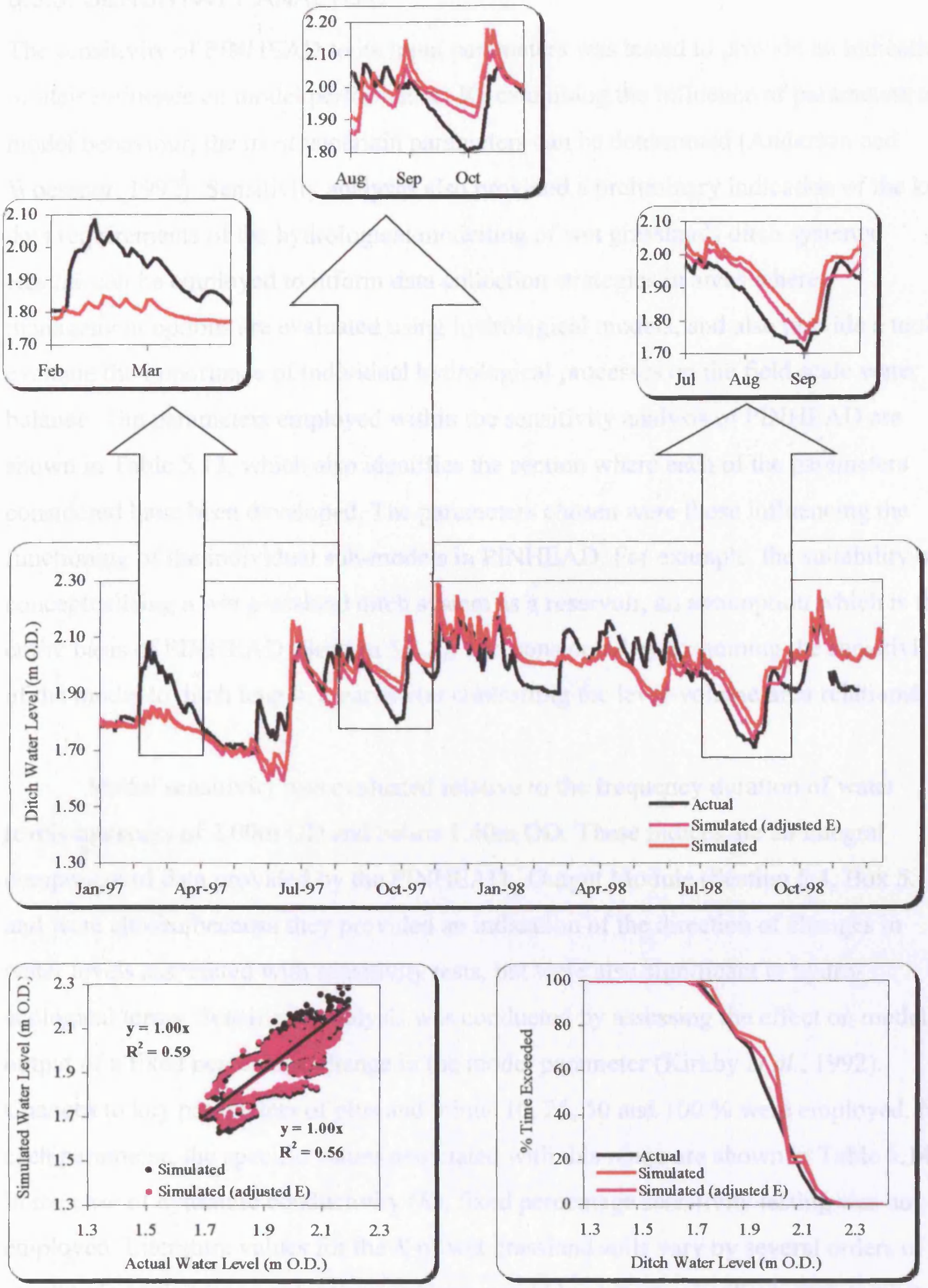


Figure 5.18. Actual and predicted water levels on the SWT Reserve during the validation period, highlighting model inaccuracies.

5.5.3. SENSITIVITY ANALYSIS

The sensitivity of PINHEAD to its input parameters was tested to provide an indication of their influence on model performance. By examining the influence of parameters on model behaviour, the most uncertain parameters can be determined (Anderson and Woessner, 1992). Sensitivity analyses also provided a preliminary indication of the key data requirements of the hydrological modelling of wet grasslands ditch systems. Results can be employed to inform data collection strategies in areas where management options are evaluated using hydrological models, and also provide a tool to evaluate the importance of individual hydrological processes on the field scale water balance. The parameters employed within the sensitivity analysis of PINHEAD are shown in Table 5.13, which also identifies the section where each of the parameters considered have been developed. The parameters chosen were those influencing the functioning of the individual sub-models in PINHEAD. For example, the suitability of conceptualising a wet grassland ditch system as a reservoir, an assumption which is the entire basis of PINHEAD (Section 5.2.3), was considered by examining the sensitivity of the model to ditch length, a parameter controlling the level-volume-area relationship.

Model sensitivity was evaluated relative to the frequency duration of water levels in excess of 2.00m OD and below 1.40m OD. These indices are an integral component of data provided by the PINHEAD_ Output Module (Section 5.4, Box 5.10), and were chosen because they provided an indication of the direction of changes in water levels associated with sensitivity tests, but were also significant in hydro-ecological terms. Sensitivity analysis was conducted by assessing the effect on model output of a fixed percentage change in the model parameter (Kirkby *et al.*, 1992). Changes to key parameters of plus and minus 10, 25, 50 and 100 % were employed. For each parameter, the specific values associated with this range are shown in Table 5.14. In the case of hydraulic conductivity (K), fixed percentage sensitivity testing was not employed. Literature values for the K of wet grassland soils vary by several orders of magnitude so that fixed percentage analysis would have been unrealistic with respect to the range of variation observed in the field (Kirkby *et al.*, 1992). As a result, the range of published values for peat and clay soils shown in Table 5.15 was employed to establish model sensitivity to K estimates.

Parameter	Notes	Module	Sub-model	Method
Slope of relationship between 30 day preceding rainfall and the runoff coefficient	See Figure 5.15 and Equation 5.30	PINHEAD_Parameters_Input	Runoff	Fixed Percentage
Rate of sluice discharge per unit head	See Equations 5.18 and Figure 5.12	PINHEAD_Parameters_Input	Sluice discharge	Fixed Percentage
C	The multiplication constant representing the assumption that the area of exchange in Darcy's Law is proportional to the hydraulic gradient	PINHEAD_Parameters_Input	Ground-surface water interactions	Fixed Percentage
Hydraulic Conductivity (*K*)	See Equation 5.13	PINHEAD_Parameters_Input	Ground-surface water interactions	Published values
Catchment Area (C_A)	A means of testing the importance of catchment delineation (Section 5.2.4)	PINHEAD_Parameters_Input	Runoff	Fixed percentage
Ditch Length (*L*)	Affecting all sub-models by its influence on the level-volume-area relationship	PINHEAD_Level_Volume_Input	All	Fixed percentage
Rainfall	Necessary time-series data	PINHEAD_Sub-model control	Rainfall, Runoff	Fixed Percentage
Evaporation	Necessary time-series data	PINHEAD_Sub-model control	Evaporation	Fixed Percentage

Table 5.13. PINHEAD parameters for which sensitivity analysis was undertaken. Parameters have been chosen due to their influence on the functioning of the individual sub-models, but also to test the theoretical underpinnings of the model.

Parameter	-100%	-50%	-25%	-10%	0%	+10%	+25%	+50%	+100%
Slope of the rainfall-runoff relationship	0	0.13	0.27	0.32	0.36	0.40	0.49	0.41	0.72
Rate of sluice discharge per unit head	0	0.063	0.031	0.113	0.125	0.138	0.156	0.188	0.250
Hydraulic Conductivity (md^{-1})	0	0.029	0.015	0.052	0.058	0.064	0.073	0.087	0.116
Catchment Area (m^2)	0	246,070	369,105	442,926	492,140	541,354	615,175	738,210	984,280
Ditch Length (m)	0	2,781	4,171	5,005	5,561	5,561	6,951	8,342	11,122

Table 5. 14. Range of values used for the analysis of the sensitivity of the PINHEAD model to its parameters. Only values associated with the fixed percentage method are shown.

Location	K (md^{-1})	Notes
Norfolk Broads, Norfolk	48.45	Max. for six fields
	0.10	Max. for six fields
North Kent Marshes, Kent	2.03 x 10^{-9}	Mean of four samples
Halvergate marshes, Norfolk	100	Surface
	0.10	1m below ground
Narborough Bog, Leicestershire	0.30	Woody peat
	0.10	Silty clay
	10	*Phragmites* peat
Wicken Fen, Cambridgeshire	0.003	-
Somerset Levels	0.75	Minimum
	1.12	Maximum

Table 5.15. Summary of the range of values presented by various authors for the hydraulic conductivity (K) of marshland soils in the UK (see Table 1.8).

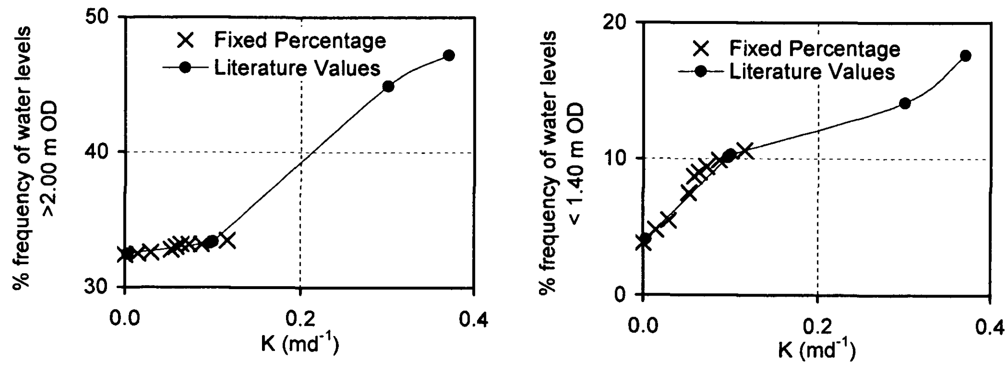


Figure 5.19. Sensitivity of PINHEAD to hydraulic conductivity.

The sensitivity of PINHEAD to various estimates of K is shown in Figure 5.19. For comparative purposes, the results of sensitivity analysis based on both published values and values obtained from the fixed percentage approach are shown in that figure. At values of K greater than $0.36md^{-1}$ the model became unstable. This threshold could not have been ascertained based on fixed percentage sensitivity testing, supporting the use of a broad range of values where these data are available. For parameters considered using fixed percentage analysis alone, results are summarised in Figure 5.20. In all cases, results are presented as a function of the frequency-duration of water levels greater than 2.00 m OD and less than 1.40 m OD. Results obtained are grouped according to the way they influence the model, or based on the methods employed for their development. For example, parameters such as the slope of the relationship between 30 day preceding rainfall and the runoff coefficient, and the rate of sluice discharge per unit head are shown together (both have been established from water level recorder charts), as are the catchment area and ditch length, established from maps.

PINHEAD was most sensitive to variations in ditch length (and therefore the level-volume-area relationship) and the rate of sluice discharge per unit head. In both cases, variations greater than plus or minus 10% resulted in large changes to the frequency-duration of water level indices (Figure 5.20), created water level trends that deviated considerably from observed water levels, and tended to cause major instabilities in the model (Figure 5.21). In the case of ditch length, the greatest instabilities were associated with increases in the parameter value. Reductions in ditch length served only to limit the replication of hydrograph peaks, as illustrated by the general decline of water levels in excess of 2.00 m OD with decreasing values of L (Figure 5.20), although effects on the seasonal variation in water levels were limited (Figure 5.21). The reverse was the case for sluice discharge. Reductions in the controlling parameter caused the greatest effects on model predictions. Increases suppressed hydrograph peaks, although the effects were not of the same magnitude as those associated with reduced sluice discharge. For increases less than +50%, the effects on the replication of water levels from June 1997 onwards were negligible because, following the re-profiling of sluice P26 (Section 5.3.4.3), water levels on the Reserve rarely exceeded the sluice level.

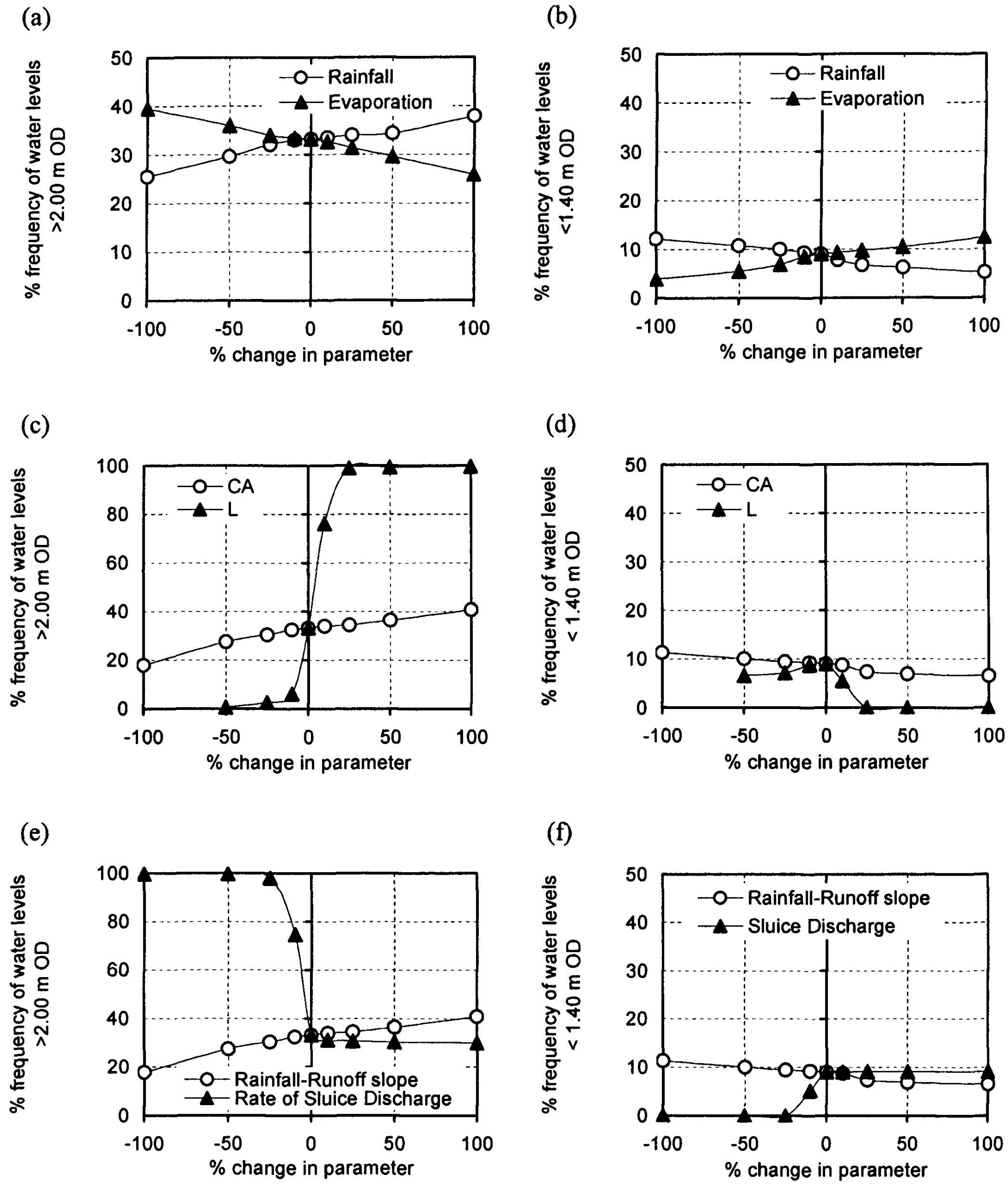


Figure 5.20. Sensitivity of PINHEAD to parameters relative to the frequency of water levels greater than 2.00m OD and less than 1.40m OD.

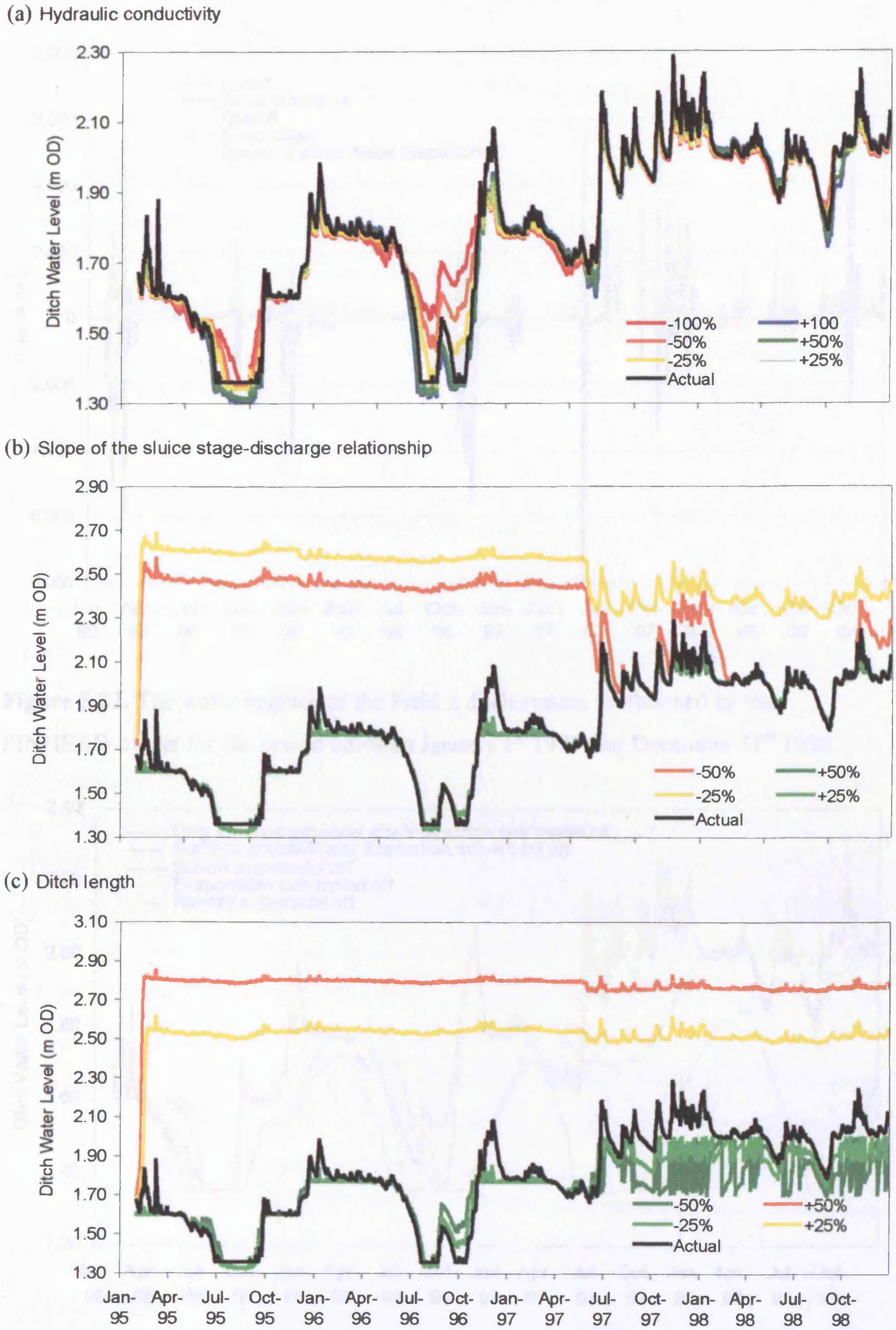


Figure 5.21. Ditch water levels on the SWT Reserve as predicted by PINHEAD based on fixed-percentage sensitivity testing of a number of key model parameters.

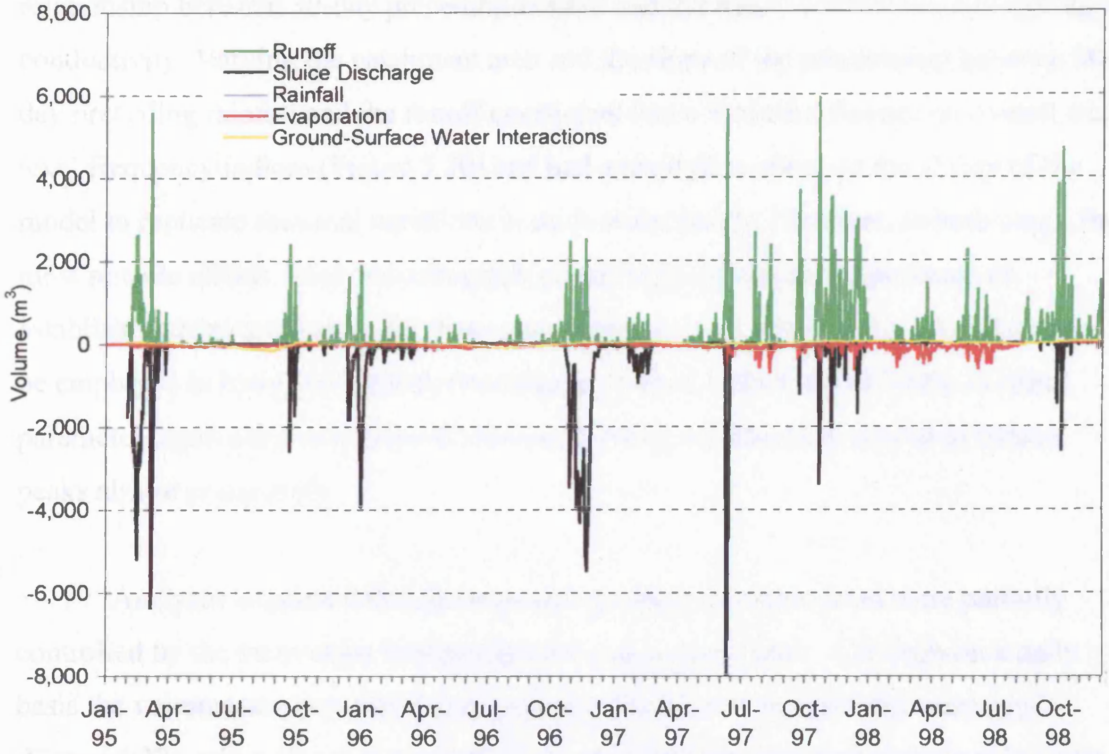


Figure 5.22. The water balance of the Field 2 ditch system as obtained by the PINHEAD model for the period between January 1st 1995 and December 31st 1998.

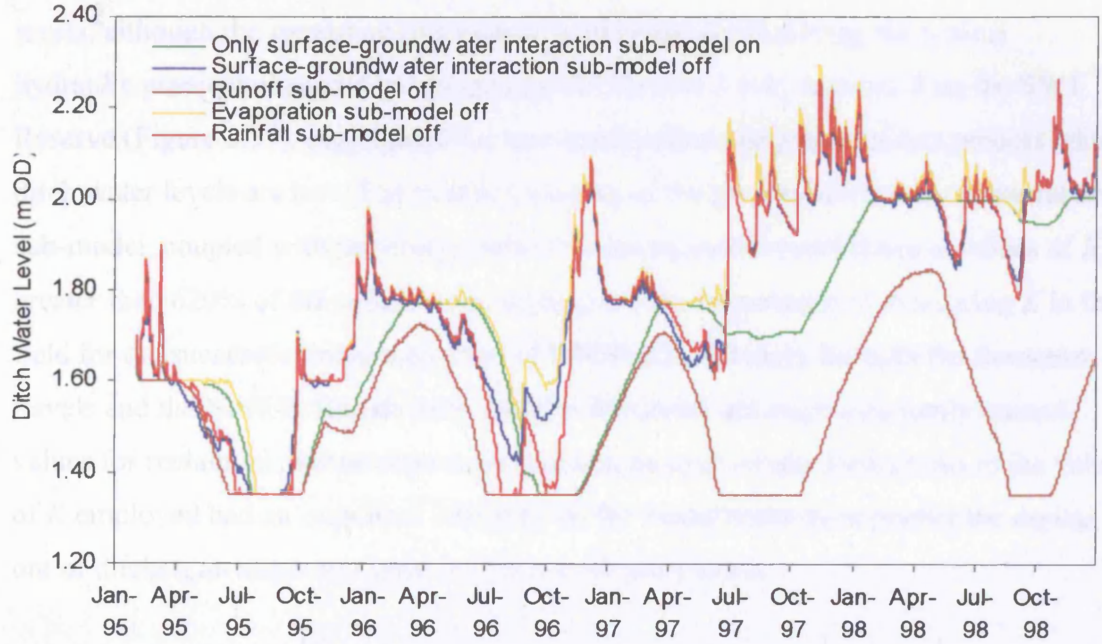


Figure 5.23. Water levels on the SWT Reserve as predicted by PINHEAD with different sub-models switched off.

The model was least sensitive to variations in catchment area, the slope of the relationship between 30-day preceding rainfall and the runoff coefficient, and hydraulic conductivity. Varying the catchment area and the slope of the relationship between 30-day preceding rainfall and the runoff coefficient had a limited influence on overall water level frequency indices (Figure 5.20) and had a negligible effect on the ability of the model to replicate seasonal variations in ditch water levels. However, in both cases, the most notable effects were on hydrograph peaks, highlighting the importance of establishing adequate values for these parameters in areas where water level data may be employed in hydro-ecological assessments. Indeed, reductions of 100% in either parameter, equivalent to the runoff sub-model being switched off, served to reduce peaks almost completely.

Analyses suggested that the seasonal variations in water level were partially controlled by the interaction between ground and surface water. Although on a daily basis the volumetric contributions of ground-surface water interactions were small (Figure 5.22), when all other processes were switched off, seasonal variations in water levels were closely replicated (Figure 5.23). Water level recessions in the summers of 1995 and 1996 were closely replicated by the model. This illustrated the limited influence of evaporation directly from the ditch water surface at the lowest ditch water levels, although the important role played by evaporation in driving the typical hydraulic gradient observed in wet grasslands (Section 1.6.4), replicated on the SWT Reserve (Figure 3.37), highlighted the important indirect influence of this process when ditch water levels are low. The overall influence of the ground-surface water interaction sub-model, coupled with previous results concerning model instabilities at values of K greater than 620% of the actual value, highlighted the importance of measuring K in the field for the successful implementation of PINHEAD. K values for both the Somerset Levels and the Norfolk Broads exceeded this threshold, although commonly quoted values for reclaimed marine clays were found to be appropriate. Reductions in the value of K employed had an important influence on the model's ability to predict the drying out of ditches, an important issue in hydro-ecological terms.

5.5.4. THE DATA REQUIREMENTS OF HYDROLOGICAL MODELLING

Sensitivity analysis has provided a preliminary indication of the data requirements of hydrological modelling in wet grassland ditch systems. In particular, the sensitivity of the model to variations in the level-volume-area relationship and sluice discharge has highlighted the need for the collection of field-based data to establish both parameters. For the establishment of the level-volume-area relationship, this should include the detailed delineation of the target ditch network and the measurement of the cross-sectional dimensions of the ditch system at a number of locations. Results suggest the inadequacy of employing the ditch dimensions proposed by Newbold *et al.* (1989) to establish these relationships, at least where they are applied within operational hydrological models. This is especially the case since the collection of cross-sectional data is a relatively simple task, and may be unnecessary if data describing ditch dimensions are available in Internal Drainage Board plans.

For sluices associated with the target ditch system, the establishment of stage-discharge relationships based on flow measurement under a variety of hydraulic conditions will also be an important component of model implementation. The sensitivity of PINHEAD to sluice discharge parameters highlights the need for a detailed evaluation of the methods employed for the estimation of flow over these structures. Results presented in Section 5.3.4.2 have provided an indication of the inadequacy of using equations for rectangular weirs for simulating discharge through sluice structures. However, this is thought to be more closely associated with the need to establish values for the influence of macrophytic vegetation on intra-channel conveyance rather than the inadequacy of the equations themselves.

Similar conclusions can be drawn with respect to the importance of methods for the estimation of runoff magnitude. Although variations in the parameters controlling runoff (catchment area and the slope of the relationship between 30 day preceding rainfall and the runoff coefficient) have a limited influence on the overall replication of inter-annual water level variability, the considerable effects on the accurate replication of hydrograph peaks suggest that a lack of suitable data may impair the model's ability to evaluate the impacts of raising ditch water levels on agricultural stakeholders. Interestingly, sluice discharge and lowland runoff were the two processes for which the least amount of data describing the calculation of their magnitude was encountered in the literature during the completion of this thesis. The close correspondence between runoff events and periods when large volumes of water are evacuated through sluices (Figure 5.22) suggests that the development of integrated methods to quantify sluice discharge and runoff are an essential component of any future work to be conducted as an extension to this thesis. Similarly, evaporation can be identified as a focus for on-going hydrological monitoring. This is especially the case where hydrological models may be employed to simulate water level management strategies that advocate the inundation of surrounding land through higher water levels. Results presented in Chapter 4, and incorporated within the PINHEAD model (Section 5.3.2), show that higher water levels potentially lead to greater evaporative loss, and losses in excess of estimates obtained by traditional means. Consequently, this issue should be considered when establishing the sustainability of revised water level management strategies in water resource terms. A preliminary assessment has been previously conducted in Section 4.8 with respect to the catchment-scale hydrology of the Pevensey Levels wetland. This assessment is furthered in Chapter 6 with regard to the field-scale hydrology of the Pevensey Levels wetland.

Overall, the model presented in this chapter has been found to provide an adequate representation of water level fluctuations on the SWT Reserve. This is particularly the case in the context of the limited data requirements of the model, as a result of which some degree of inaccuracy was expected. Of special interest is the fact that the frequency of the range of water levels, an index of particular significance in hydro-ecological terms, is closely replicated. In Chapter 6, these data are employed to address the key water level management issues of interest to wetland stakeholders, identified during meetings of the Pevensey Levels Study Group (Section 2.8.3).

CHAPTER 6

THE IMPACTS OF WATER LEVEL MANAGEMENT ON WETLAND STAKEHOLDERS

6.1. Introduction

A large number of wetland restoration schemes are applied across wetlands in the UK. These schemes have a wide remit in terms of the management practices they prescribe, and extensive funds are dedicated to them. Chapter 1 has previously reviewed these schemes, focusing on the water level, and other prescriptions associated with each (Section 1.7). Chapter 2 has considered those which are in operation on the Pevensey Levels wetland in more detail (see Section 2.8). However, to date few quantitative evaluations of the success of these schemes in ecological terms have been made on the Pevensey Levels, nor have the impacts on agricultural productivity been considered. This is mainly a factor of the lack of tools available that enable an assessment of the effects of changes in ditch water level regimes on nature conservation or agricultural interests in an area.

Chapter 5 has described a hydrological model capable of simulating ditch water levels on the Pevensey Levels wetland. However, the hydrological model alone is insufficient to deal with the concerns of local stakeholders regarding changes to local water level management strategies. In this chapter, a modelling approach to quantify the effects of different water level prescriptions on stakeholders is discussed. This chapter describes the development of a hydro-ecological sub-model based on simple principles that links output ditch water level data from the hydrological model to quantitative ecological information describing the preferences of individual wetland species and stakeholders to different water level regimes and/or hydrological conditions. In later sections, the model developed is used to estimate the way in which different sluice settings and climatic conditions (*e.g.* climate change) affect local stakeholders, and assesses ways in which water levels may be tailored for their varying requirements.

The methodology described and the simulations conducted in this chapter have been primarily undertaken in response to the requirements of stakeholders on the Pevensey Levels. In meetings of the Pevensey Levels Study Group, composed of a cross-section of wetland stakeholders (Section 2.8.3), farmers on the Pevensey Levels have consistently highlighted the economic costs incurred as a result of restoration-orientated water level management strategies. These problems have specific implications for the continued sustainability of wetland restoration initiatives on the Pevensey Levels, and in wet grasslands in general. Similar concerns have been raised in areas of conservation importance, where changes in the water level regime of ditches may alter the composition of the fauna and flora (RSPB *et al.*, 1997).

The primary objective of the methods employed in this chapter is to provide a means of quantifying the impacts of changes to ditch water level management in an area prior to the implementation of a scheme. The limited funding for capital works associated with WLMPs means that, where possible, water level objectives must be met using the existing drainage network. However, higher water levels for conservation may not always be achievable in a drainage system designed for intensive agriculture. For example, existing maximum sluice levels may not allow the attainment of the higher target water levels associated with nature conservation requirements. The method therefore also has the added objective of being capable of providing an evaluation of the potential to deliver water level objectives within the existing drainage infra-structure, and the identification of key actions required to meet pre-defined water level objectives.

The method presented also has potential as a means of designing the specific water level regimes associated with Water Level Management Plans, which seek to *'integrate the water level requirements of conservation, flood defence and agriculture'* (MAFF, 1994). Approaches for the implementation of WLMPs on the Pevensey Levels have been discussed in Section 2.8.2, but to date the water level regimes to be applied have not been determined, although some integrative water level regimes for wet grassland have been proposed (*eg* Spoor and Gowing, 1993; Figure 1.19). Guidelines set out by MAFF (now DEFRA) clearly state that WLMPs should be designed based on the stakeholders in a locality and their requirements (MAFF, 1994). The approach described in this chapter aims to provide a tool to satisfy this requirement.

Flexibility will be an important component in the development of WLMPs. For example, ditch water level targets will have to be established based on the agricultural activities practiced in the target area and the species of nature conservation importance that are present. In arable areas, an important influence on the water level regime adopted will be the crops grown (Table 1.12) and the nature of the substrate (Figure 1.13). In nature conservation terms, management regimes adopted on the Somerset Levels and the North Kent Marshes for the benefit of wet grassland bird species will not necessarily be suitable on the Pevensey Levels, where the flora and fauna of national biodiversity interest inhabit the ditches (see Section 2.6). The design of integrative water level regimes which seek to unite the requirements of agriculture and nature conservation requires a different approach to the '*install it and forget it*' mentality (Skaggs, 1992), incorporating flexibility to alter sluice levels in response to environmental conditions that may affect one or another of the stakeholders included in the WLMP. This may be necessary following periods of prolonged rainfall during the summer, which might impact upon ground-nesting birds, or during the winter to avoid extensive damage to agricultural crops or grazing interests. The hydro-ecological sub-model described in this chapter is capable of identifying the times of year or environmental conditions that will make this type of mitigation necessary.

In this chapter, the functioning and physical bases of the hydro-ecological sub-model are described. The sub-model is then applied to address four specific objectives of importance in the local area. The model is firstly use to explain the current biodiversity status of the Pevensey Levels wetland. The second objective is to evaluate the suitability of water level prescriptions associated with existing wetland restoration schemes (ESA, WES, Countryside Stewardship) in terms of agriculture and nature conservation. Thirdly, the model is used to investigate the potential for providing the water level targets associated with various wetland restoration schemes under scenarios commonly quoted in the climate change debate. Finally, this chapter discusses the possibility of designing a water level management regime that integrates the requirements of all stakeholders on the Pevensey Levels and which can be applied as part of the WLMPs for the site. The assessments presented have been conducted on the Sussex Wildlife Trust Reserve, the area for which the hydrological model presented in Chapter 5 was developed, and where the extensive field scale data reviewed in Section 3.6 was collected.

6.2. Hydro-ecological modeling in PINHEAD

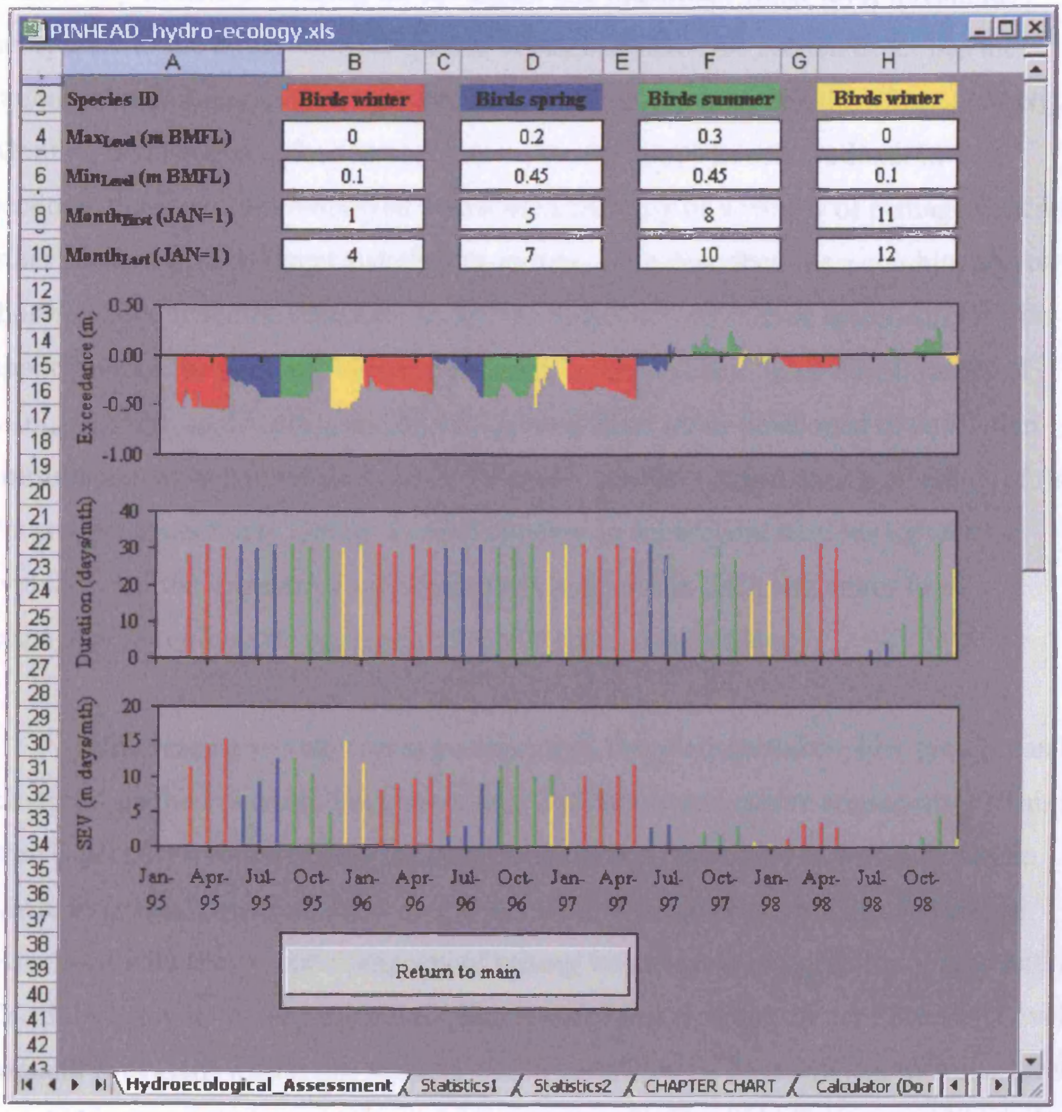
In PINHEAD, the impacts of different water level regimes on local stakeholders can be evaluated using the PINHEAD_Hydroecology Module. For the operation of the sub-model, a value of mean field level is required and is input in the PINHEAD_Parameters Module in the main model screen (Box 5.7). The sub-model incorporates the assumption that the most discriminating variable determining the presence or absence of a species in a wetland is the magnitude and duration of hydrological conditions that fall below or rise above species requirements (Gowing *et al.*, 1999). The sub-model adopts the 'Sum Exceedence Value' (SEV) approach, advocated by Gowing *et al.* (1999), who have employed the approach to explain the distribution of floral communities on the Somerset Levels in areas where water level management history is well-established. The SEV method assumes that a given species has characteristic maximum and minimum threshold water levels and that beyond these, the species is subjected to physiological stresses that may reduce its abundance over an area. For plants, these physiological stresses are both direct and indirect. Inundation, drought or the water level regime may influence water and oxygen supply to the roots and be indirectly associated with soil nutrient availability, soil temperatures and sward management (Gowing and Spoor, 1998).

The main difference between the use of the SEV method on the Pevensey Levels relative to previous applications, is its application to assess the suitability of ditch water level regimes. Although ditch water level targets are a typical focus of wetland restoration strategies applied on wet grassland areas in the UK, previous assessments have tended to consider only the impacts of water table level variations on biota inhabiting the field surfaces. However, in the case of the Pevensey Levels, the drainage channels and rhymes connected to them are far more important for nationally rare and scarce plant species than the fields (Section 2.6). The development of the PINHEAD_Hydroecology Module therefore compliments previous hydro-ecological work conducted in wet grassland habitats. This is particularly the case with respect to the development of data describing the water level requirements of agricultural stakeholders, which have been established during discussions with local farmers. To date, few data are available in the literature describing the water level requirements of different agricultural practices: Figure 1.11 has collated all the data available to the best of the author's knowledge.

The PINHEAD_Hydroecology Module is shown in Box 6.1 and works by linking simulated ditch water level data from the PINHEAD hydrological model to information describing the hydrological preferences of a wetland stakeholder. These data take the form of quantitative data describing the maximum and minimum threshold water levels at different times of year (termed Min_{level} and Max_{level} respectively). For agricultural and nature conservation stakeholders on the Pevensey Levels wetland, the development of values of Min_{level} and Max_{level} is described in Section 6.3. In all cases, values of Min_{level} and Max_{level} have been established as a function of the mean field level, mainly because target water levels associated with existing wet grassland restoration schemes are expressed in these terms (Table 1.15, Table 1.18; Table 2.13). Other input data required by the PINHEAD_Hydroecology Module are the first and last months during which these thresholds are effective (termed $Month_{1st}$ and $Month_{Last}$ respectively), where January = 1and December = 12.

Within the PINHEAD_Hydroecology Module, the suitability of a specific water level regime on up to four species, or on the same species at four different times of year, can be tested simultaneously. If the simulated ditch water level during the period of interest specified is greater than the value of maximum water level threshold (Max_{level}), a positive exceedence is recorded. If the water level drops below the minimum water level threshold (Min_{level}), a negative exceedence value is reported. Output data from the PINHEAD_Hydroecology Module takes the form of three graphs quantifying:

- the extent to which the ditch water level varies beyond the preferences of the species on a daily basis, termed the exceedence (m),
- the duration of any exceedence (exceedence duration) expressed in days per month, and
- the 'Sum Exceedence Value' (SEV), the product of exceedence and exceedence duration, expressed in metre-days per month.



Box 6.1. The PINHEAD_Hydroecology Module. The colours for exceedence, exceedence duration and sum exceedence value charts identify either different species, or the requirements of an individual species at different times of year. Colours shown in the chart are equivalent to the colours in the 'Species ID' cells.

6.3. Stakeholder water level requirements

The successful implementation of the PINHEAD hydrological model to provide meaningful hydro-ecological assessments of ditch water level regimes relies on the availability of data describing the minimum and maximum water level thresholds acceptable to the target wetland species or stakeholder. This section describes the rationale and methodology employed to establish data describing the Min_{Level}, Max_{Level}, $Month_{1st}$ and $Month_{Last}$ for stakeholders on the Pevensey Levels wetland. In later sections, these data are employed to test the suitability of a variety of management and other scenarios on different stakeholder groups. Data described are a combination of those collated from the literature, as well as thresholds developed specifically for the application of the approach on the Pevensey Levels wetland. In all cases, values of Min_{Level}, Max_{Level}, $Month_{1st}$ and $Month_{Last}$ have been either developed or verified in consultation with stakeholders on the Pevensey Levels wetland during meetings of the Pevensey Levels Study Group. Scenarios tested in subsequent sections include the evaluation of the impacts of various historic and current ditch and sluice level management regimes on the requirements of these stakeholders.

With respect to water level management, two distinct stakeholder groups can be identified on the Pevensey Levels wetland: agriculture and nature conservation. This has been especially evident during the discussions of the Pevensey Levels Study Group where in general, conflicts regarding water level management strategies have been associated with the potential impacts of raising water levels on agricultural productivity. The dichotomy in the water level requirements of stakeholders on the Pevensey Levels wetland is evident within this section, which describes the development of water level thresholds for agriculture and nature conservation. Further sub-divisions within each of the sections address the contrasting water level requirements of different stakeholder groups. For example, species of nature conservation importance on the Pevensey Levels can be sub-divided into three main groups: birds (waders and anatids), flora and invertebrates (*e.g. Dolomedes plantarius*). Agricultural practices on the wetland include arable agriculture, as well as intensive and extensive grazing (Section 2.2.2). The structure adopted to describe the development of water level threshold data for different stakeholder groups as required by the PINHEAD_Hydroecology Module reflects these differences.

6.3.1. FLORA

Detailed data describing the hydrological preferences of a broad range of wetland vegetation are provided in a report entitled 'The Water Level Requirements of Wetland Plants and Animals' (Newbold and Mountford, 1997). The report provides an indication of the maximum and minimum thresholds for a variety of wetland floral and faunal species, established in habitats where limited changes in the water level regime have occurred through time (Spoor *et al.*, 1992). A sample data set from this report has been shown in Table 1.5, describing the water level requirements of a variety of nationally rare and scarce floral species of wet grassland. A community-based approach has been adopted by RSPB *et al.* (1997), who provide a qualitative classification of suitable inundation frequency regimes on characteristic vegetation assemblages of wet grassland (Table 6.1), based on the classification of Rodwell *et al.* (1992).

For the Pevensey Levels, the characteristic floral assemblages of ditches and fields have been described in Section 2.6. The species chosen for use within subsequent simulations were those present on the wetland classified as nationally rare or scarce. Nationally rare species are those which occur in 1-15 10x10km grid squares in the UK, and nationally scarce species are those found in 16-100 grid squares (Perring and Walters, 1962; Preston and Croft, 1997). The species employed were therefore the sharp-leaved pondweed, *Potomageton acutifolius* (Nationally Rare), Water soldier, *Stratiotes aloides* (Nationally Scarce), Hair-like pondweed (*Potomageton trichoides)*, Greater water-parsnip (*Sium Latifolium)*, Bladderwort (*Utricularia* spp.), Rootless duckweed (*Wolfia Arrhiza)*, Water-Violet , *Hottonia palustris* (Nationally Scarce) and Soft hornwort (*Ceratophyllum submersum)*. The distribution of these species in the UK and on the Pevensey Levels wetland has been previously shown in Figures 2.13 and 2.16 respectively. The choice of species was based mainly on the fact that it is these species which provide the Pevensey Levels with its nationally important status, as evidenced in Ramsar and SSSI citations (Appendix 2.2 and 2.3). All the rare and scarce floral species identified on the wetland are aquatic and inhabit the ditches, further supporting the use of a ditch model to evaluate the potential impacts of a variety of management options upon them. For example, the model is used to explain the distribution of the species on the wetland, especially their concentration in un-drained areas used primarily for grazing relative to pump-drained areas.

Community	Flooding regime
MG4*	Winter flooding occasionally persisting into the spring
MG5	Normally none, standing water in winter is normally associated with other types
MG6	No flooding- or only in very exceptional years
MG7	Where flooded regularly in winter, *Lolium P.* is accompanied by meadow species of *Festuca* and *Alopecurus*
MG8*	Deliberately flooded in the past for long period in the winter and spring. This tradition is now rare and the community is found where natural floods occur by rivers
MG9	Periodically inundated, *eg.* in furrows - not flooded deliberately
MG10	Not normally flooded
MG11*	Inundated by fresh or brackish water, but also prone to periods of drying out
MG12*	Prone to inundation by brackish water, more rarely tidal water or salt spray
MG13*	Regularly flooded by fresh water - sometimes for long periods
M22*	Often flooded in winter, very variable in duration, resultant in floral variety
M23*	Not usually flooded
M24*	Very seldom flooded
M25*	Not usually flooded
S5*	Regular, very prolonged winter flooding
S6	Regular, prolonged winter flooding
S7	Regular, prolonged winter flooding
S22*	Regular, prolonged winter flooding, often through into late spring / summer

* Communities considered to be agriculturally unimproved and semi-natural in character

Table 6.1. Hydrological requirements of wet grassland vegetation communities (from RSPB *et al.*, 1997).

For the species listed, the maximum and minimum water depth thresholds proposed by Newbold and Mountford (1997) are listed in Table 6.2. These data are expressed as a function of metres below field level, which as previously stated, are the data required by the PINHEAD_Hydroecology Module. Values of Min_{Depth} and Max_{Depth} were adjusted based on the cross-sectional profiles of ditches in the area to which PINHEAD was applied. The specific cross-sectional dimensions of ditches on the SWT Reserve has been previously described in Sections 3.5.2 and Figure 3.24. The dimensions employed within the PINHEAD model have been described in Section 5.2.3. Based on the mean cross-sectional dimensions of the ditch system on the SWT Reserve, water depths of 1.00m and 0.20m, the maximum and minimum threshold water levels proposed for *Stratiotes aloides*, are afforded by water levels 0.12m and 0.92m below the mean field level. For all floral species considered in the analysis, values of Min_{Level} and Max_{Level} developed by these means are shown in Table 6.2. These data were assumed to be effective during the macrophyte growth season, which extends between April and September (Section 5.3.4.3).

The data obtained for all floral species were similar, allowing the treatment of the ditch vegetation on the Pevensey Levels as an aquatic community. Exceptions to this rule were *Sium latifolium* and *Hottonia palustris*, which can tolerate water levels below the ground surface, and therefore provided a useful indicator of the influence of water level management strategies on marginal floral communities on the Pevensey Levels wetland. The species chosen also contained two floating species, *Wolfia Arrhiza* and *Ceratophyllum submersum*, for which no water level preference data have been presented. This species could not therefore be considered within the framework developed. For the purpose of the assessment it is assumed that the water level requirements of these floating species will be similar to emergent floral species (*eg. Potomageton* spp., *Stratiotes aloides*) because, on the Pevensey Levels, rare floating species are commonly found in association with other rare marginal and emergent floral species (Neil Fletcher, SWT Reserves Officer, Pers. Comm.; Figure 2.16).

Species	Max_{Depth} Newbold and Mountford (1997) (m)	Min_{Depth} Newbold and Mountford (1997) (m)	Max_{Level} Adjusted for PINHEAD (m BMFL)	Min_{Level} Adjusted for PINHEAD (m BMFL)
Sium latifolium	0.40	-0.30*	0.72	1.42*
Urticularia sp	1.00	0.10	0.12	1.22
Wollfia arrhiza	Floating species	Floating species	Floating species	Floating species
Hottonia palustris	0.80	-0.05*	0.32	1.17*
Ceratophyllum submersum	Submerged floating species	Submerged floating species	Submerged floating species	Submerged floating species
Potomageton acutifolius	0.80	0.10	0.32	0.92
Potomageton trichoides	1.00	0.20	0.12	0.92
Stratiotes aloides	1.00	0.20	0.12	0.92

* indicates that the species is tolerant to water levels falling below its root base and therefore likely to be a marginal species inhabiting the ditch bank.

Table 6.2. Water level requirements of floral species of the Pevensey Levels wetland. Data are applicable during the macrophyte growth season (April-September). Data have been adjusted based on descriptions of the cross-sectional dimensions of ditches on the SWT Reserve so that they can be expressed as a function of water levels below mean field level (m BMFL).

6.3.2 THE FEN RAFT SPIDER

In national terms, the fen raft spider, *Dolomedes plantarius*, can be regarded as the flagship species of nature conservation importance on the Pevensey Levels wetland. This species is only present on one other site in the UK, Redgrave and Lopham Fen, Suffolk (Section 2.6), although the population at Pevensey is considerably larger, consisting of over 1200 individuals (Jones, 1992). These factors make the impacts of proposed water level management strategies on the species an important consideration. However, few quantitative data describing the water level requirements of the fen raft spider are available, although management in Redgrave and Lopham Fen to enhance the habitat of this species has focused on the provision of 'deep pools and ponds' (Daily Telegraph, 07/08/1991).

The most detailed study of the habitat requirements of the fen raft spider is a study on the Pevensey Levels by Jones (1992), previously discussed in Section 2.6. The ideal habitat characteristics of the species are summarized in Table 6.3 and have been adapted for application within PINHEAD. Based on the conclusions provided by the study, a maximum tolerable water level (Max_{level}) equivalent with field level can be used throughout the year, as the spider is not found in parts of the grazing marsh that are temporarily flooded (Jones, 1992). The importance of *Stratiotes aloides* to the species for breeding and as a habitat also allow the use of threshold data developed for this floral species (Section 6.3.1, Table 6.2) as a surrogate for the suitability of different water level management regimes on the species. These latter data were employed to establish a value of Min_{Level} for *Dolomedes plantarius*. An alternative approach was the development of a threshold based on the requirement of 'constant' water levels throughout the year suggested by Jones (1992) (Table 6.3). This however was considered an excessively subjective approach, due to the difficulty of defining the term 'constant water levels'. In adopting a value of Min_{level} equivalent to that of *Stratiotes aloides*, the requirement of permanent standing water (Table 6.3) was also satisfied. The value of Min_{Depth} for *Stratiotes aloides* is 0.25m (Newbold and Mountford, 1997), simulating the requirement that ditches should never run dry (spiders have however been observed in near-dry ditches during the summer of 1993). Values of Min_{Level} and Max_{Level} were applicable all year round as the species is reliant on ditches for breeding and feeding, and hibernates in tussocks close to water margins (Jones, 1992).

Habitat requirements of the Fen Raft Spider *Dolomedes plantarius*
• Open sunny location, wide ditches or ditches with few densely vegetated banks
• Permanent standing water near bank surfaces, but no flooding
• Marginal bank vegetation <1 m tall, especially *Juncus spp.* and *Carex spp.*
• Floating vegetation, especially *Hydrocharis* and emergents, especially *Stratiotes aloides*

Table 6.3. The habitat requirements of *Dolomedes Plantarius* (from Jones, 1992).

Species	Maximum water depth (m)	Source of data
Anatids	<0.5	Thomas, 1982
Pintail	<0.45	Thomas, 1982
Teal	<0.25	Thomas, 1982
Teal	<0.2	Newbold and Mountford, 1997
Shoveler	<0.3	Newbold and Mountford, 1997
Mallard	<0.35	Newbold and Mountford, 1997
Swans	<1.00	Newbold and Mountford, 1997
Tufted duck	>2.00	RSPB *et al.*, 1997
Pochard	>2.00	RSPB *et al.*, 1997

Table 6.4. Water level requirements of selected wildfowl.

6.3.3 BIRDS

The precise water level requirements of the characteristic waders of wet grassland wetlands are intrinsically linked to the physical characteristics of the substrate that dictate the soils' susceptibility to probing by wetland birds. Flooding in winter and early spring is an over-riding requirement, providing food by releasing seeds trapped in vegetation, flushing invertebrates out into the open, and softening the soil (RSPB *et al.*, 1997). Observations of the distributions of characteristic wet grassland bird species (lapwing, redshank and black-tailed godwit) on the Ouse Washes have indicated that all three tend to nest and feed around pools of open water (O`Brien, 1998). For most anatid species, large areas of surface water with an average depth of less than 0.5m are also required (Thomas, 1982). An indication of the range of depths favoured by individual anatid species is provided in Table 6.4. In general therefore, water level management for both waders and anatids on wet grasslands relies on the provision of extensive inundated areas, although a crucial aspect of this management is a seasonal approach to water level management. For example, inundation during the fledging of ground-nesting wading birds can negatively impact upon the populations of those species. Such aspects are reflected in the thresholds developed for application within the PINHEAD_Hydroecology sub-model.

To create the ideal water table regimes required, RSPB *et al.* (1997) suggest water levels providing inundation of 30-60 % of the target site to a water depth less than 0.2 m between December and March, declining to 20% of the target area by April. Based on these requirements, target water levels for birds on the SWT Reserve were established using the Digital Elevation Model shown in Figure 5.6 to quantify the areal extent and depth of inundation at a variety of ditch water levels. For the SWT Reserve, flooding extent and depth at a variety of ditch water levels is shown in Figure 6.1. Based on these data, the water level required between December and March to satisfy suggestions by RSPB *et al.* (1997) was approximately equivalent to 2.30m OD, or 0.08m above the mean field level (AMFL). During April, a water level of 2.25 m OD (0.03m AMFL) was required. These data were those employed as values of Min_{Level} during appropriate months. For Max_{Level} during the equivalent period, a water level associated with 80% of the Reserve being inundated was adopted (2.50 m OD). This ensured that some dry areas would remain for roosting.

In contrast, thresholds developed for the period between late spring and early autumn, sought to highlight the need to limit inundation that might reduce the food supply. O'Brien (1998) for example has reported declines in the numbers of black-tailed godwit on the Ouse Washes due to spring inundation. Earthworms cannot withstand prolonged flooding (Newbold *et al.,* 1989), although they can survive in areas where the water table is high since 75 % of earthworm biomass is found in the top 0.04m of the soil (Voisin, 1959). Spoor and Gowing (1995) suggest a value of 0.35m below mean field level (BMFL) as appropriate for this food source. Summer flooding can also have detrimental impacts on larval stages of aquatic insects: in September it can damage populations of non-mobile terrestrial invertebrates and inundation before the end of this October will prevent beetle and cranefly species from laying eggs (RSPB *et al*., 1997).

For both feeding and nesting, different wading birds favour different vegetation canopy types (Table 6.5). In terms of hydrological conditions, RSPB *et al.* (1997) suggest an ideal depth of water table to be 0.00m in May, falling to 0.20m below field level for May and June and to 0.50m below field level by mid-July. During May the value of Max_{Level} employed was therefore no higher than the mean field level and receded to 0.20m from the field surface by July. Between August and the end of October a water level of 0.45 m below the field surface was employed as the value of Max_{Level}. The use of this value is related to calculations conducted in Section 6.3.2.3 that describes a calculation to establish the water level that would be required to absorb the rainfall associated with a 1 in 3 year storm for the development of thresholds associated with grazing. This water level would also potentially limit the ditch induced flooding that could impact upon the invertebrate prey of wading birds.

The establishment of the value of Min_{Level} for the equivalent period was based on suggestions by RSPB *et al.* (1997), who state that 'for wet grassland birds, the ditch water level between May and July should not fall below 0.45 m of the field surface'. Between August and October, a value of Min_{Level} accordant with the provision of at least 0.30m of water in ditches (0.82m BMFL) was employed. This value was somewhat notional, reflecting the fact that in wet grasslands, this is the period of least interest in terms of birdlife since most breeding species fledge by June (Burgess and Hirons, 1990). For the entire year, the values of Max_{Level} and Min_{Level} developed for wet grassland birds are illustrated graphically in Figure 6.2.

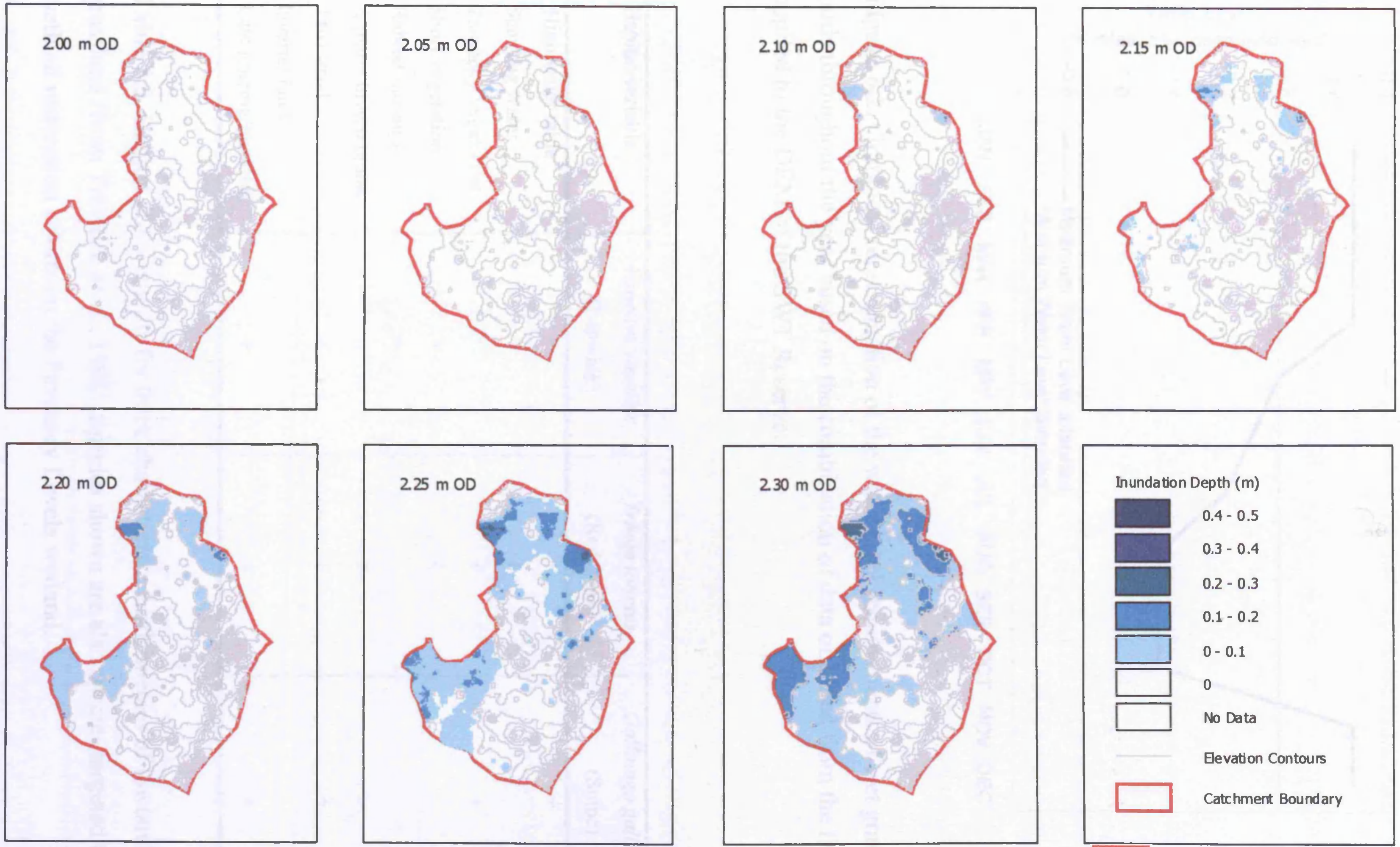


Figure 6. 1. Inundation extent and depth on the SWT Reserve at a variety of ditch water levels.

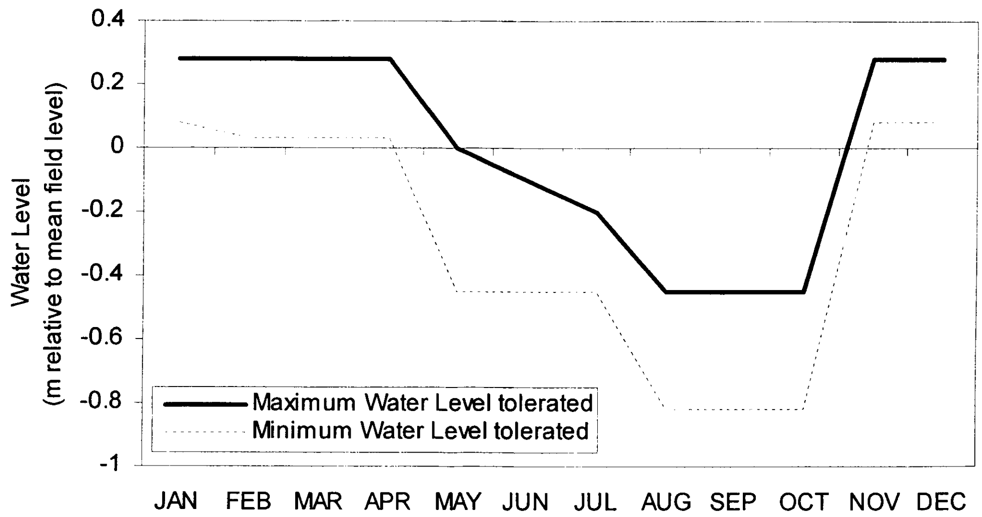


Figure 6.2. Graphical representation of the water level requirements of wet grassland birds throughout the year based on the combination of data obtained from the literature applied to the DEM of the SWT Reserve.

Habitat variable	*Vanellus vanellus* (Lapwing)	*Tringa totanus* (Redshank)	*Gallinago gallinago* (Snipe)
High water table	+	+	+
Standing water	+	+	
Tussocky vegetation		+	+
Short vegetation	+	+	
Habitat mosaics	+		
Aquatic invertebrates		+	
Terrestrial invertebrates	+	+	+
Late grazing/mowing	+	+	+

Table 6.5. Habitat requirements for three characteristic birds species of lowland wet grassland (from Treweek *et al.*, 1998). Species shown are also species targeted by wetland restoration efforts on the Pevensey Levels wetland.

6.3.4. AGRICULTURE

6.3.4.1. Arable farming

For arable agriculture, the development of values of Max_{Level} and Min_{Level} was based on the objectives of water level management in arable areas that can be summarised as:

- limiting waterlogging and flooding,
- providing sufficient crop irrigation at times of high evaporative demand, and
- ensuring access to the land and the workability of the soil.

The greatest cause of decreased production in agricultural systems is waterlogging (Garcia *et al.,* 1992), although the impact is dependant on the timing of the event relative to the crop development stage (Smedema and Rycroft, 1983) (Figure 6.3) and the cumulative duration of the event (Mann and Green, 1985). Metabolically active plant tissue will die within a few days if oxygen is excluded (Gowing and Spoor, 1998). Indirect effects are similarly important. Limiting waterlogging ensures the soil is kept well aerated for crop growth in the following season. Working the soil when it is too wet can also result in considerable yield loss in subsequent crops. Increases in bulk density when the soil is trafficked at high water contents are common, affecting the timing of planting and harvesting (Oskoui, 1992). The potential impacts of waterlogging on farming on the Pevensey Levels can be illustrated using the data provided by Jarvis *et al.* (1984) who identify a reduction of 31 days in the number of work days between September and April in a wet year relative to an average year.

The ideal water table conditions for arable agriculture as proposed in the literature are reviewed in Table 6.6. An especially high number of these describe arable cropping in the Netherlands where climatic conditions can be considered analogous to the UK. To achieve the ideal water table conditions, Van Bakkel (1988) suggests a water level of 1.45 m BMFL between October and February and 0.9m BMFL during the spring and summer (Figure 1.11). Slightly higher water levels, 1.1m BMFL in winter and 0.5m BMFL in summer, are proposed by Ritzema (1994). Similarly, for ditch system with a total length of 5100 m and a wet cross-section of 4.2 m, similar to the dimensions of the ditch system modelled, Smedema and Rycroft (1983) propose a winter ditch water level of 0.8m BMFL (Table 6.7).

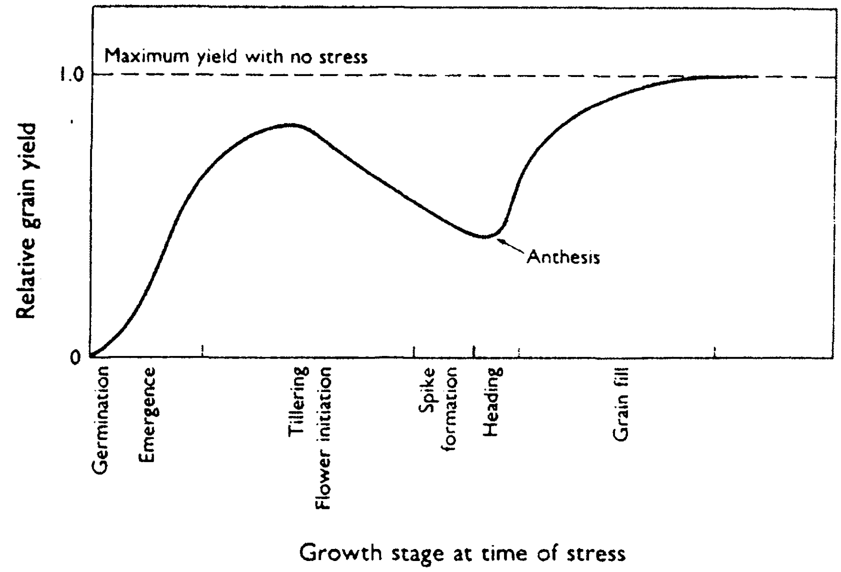


Figure 6.3. Relative grain yield of arable crops in response to water stress during crop development (from Loomis and Conor, 1992).

Author	Water table elev. (m BMFL)	Notes
Muller (1992)	0.9 – 1.1	Arable cropping (general)
Mann and Green (1978)	0.9	Arable cropping (Somerset Levels)
Cook and Moorby (1993)	0.8 – 1.0	Arable cropping (general)
Smedema and Rycroft (1983)	0.75	Arable cropping (Netherlands)
Van Bakkel (1988)	1.3 (winter), 0.8 (summer)	Arable cropping (Netherlands)

Table 6 6. The ideal water table levels for arable agriculture.

Ditch length (m)	Cross-section (m^2)	Land level (m MSL)	Water level (m MSL)	Water level (m BMFL)	Bed level (m MSL)
5100	4.2	9.3	8.5	0.8	7.5
3900	6.9	9.0	8.0	1.0	6.7
3500	10.5	8.5	7.6	0.9	6.1
2400	11.3	8.2	7.2	1.0	5.7
1000	15.9	8.0	6.8	1.2	5.0
500	16.9	8.2	6.6	1.6	4.8

Table 6.7. Standards for the design of land drainage channels (from Smedema and Rycroft, 1983).

The applicability of these data in a UK context was tested by comparison with values of the maximum tolerable water level for arable farming ($Max_{Level\ Arable}$) calculated based on the prescribed drainage design standards commonly applied in the UK. Table 1.11 has shown that in areas of the UK where cereal crops are grown, the winter design drainage standard is a 1 in 10 year storm event. To limit waterlogging and flooding, the ditch system should therefore be capable of absorbing all the runoff from such an event. To establish $Max_{Level\ Arable}$, the 2-day M10 rainfall calculated based on the FSR procedure (NERC, 1975) was taken as a representative index of a 1 in 10 year hydrological event. For the Pevensey Levels wetland, the 2-day M10 was equivalent to 63mm, of which, based on the analysis presented in Section 5.4.4.2, a maximum of 63 % will become runoff. Coupled with information describing the extent of the Field 2 catchment, these data were employed to calculate the runoff volume that would be generated on the SWT Reserve by the 2-day M10 storm event, termed the 2-day M10 volume ($V_{2day\ M10}$)(m^3). The value of Max_{Level} could then calculated as the level equivalent of the difference between $V_{2day\ M10}$ and the bankfull storage ($V_{Bankfull}$) where:

$$V_{MAXArable} = V_{Bankfull} - V_{2day\ M10} \qquad \text{(Equation 6.1)}$$

$V_{Bankfull}$ was calculated by application of the level-volume relationship described in Section 5.2.3 to a level of 2.00m OD, the inundation threshold level (Section 5.2.5).

The ditch water level required to ensure that no inundation occurred during the 1 in 10 year rainfall event was calculated as 1.35 m O.D, or 0.87 m BMFL. This value was applied during the winter months, taken as the period between crop harvesting and a month after planting in the following year. Crops are particularly susceptible to waterlogging during germination (Loomis and Connor, 1992) (Figure 6.3) and the farmer will maintain water levels as low as possible during this period. The provision of low water table conditions following harvest is also essential to maximise yields in subsequent years (Bill Gower, Farmer, pers. comm.). For the two main crops grown on the Pevensey Levels wetland, barley and wheat, dates of sowing and harvest are given in Table 6.8. A notional value of Min_{Level} equivalent to the bed level (1.12 m BMFL) was adopted during the equivalent period reflecting the limited importance of maintaining water in the ditches during the winter months and the need to provide flood storage capacity.

Crop type	Sowing date	Harvest date
Spring Barley	March	July / August
Spring Wheat	March	July
Winter Barley	Late September / early October	August and September
Winter Wheat	Late September / early October	August and September

Table 6.8. Approximate dates of sowing and harvest for crops commonly grown on the Pevensey Levels wetland.

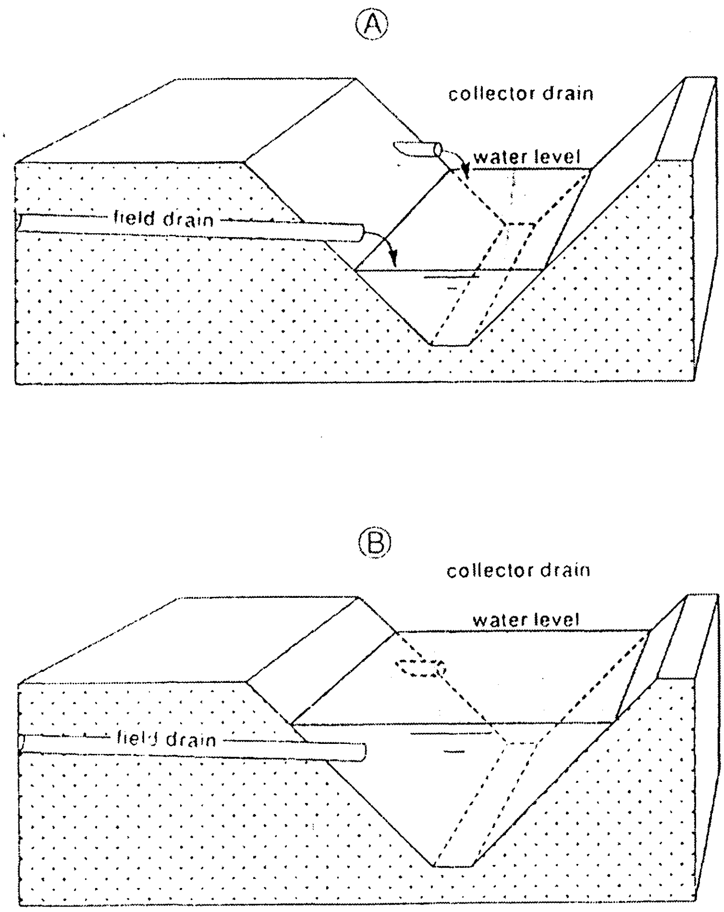


Figure 6.4. Diagrammatic representation of the potential impacts of high water levels on field under-drainage systems (from Ritzema *et al.*, 1996).

Drought stress can cause similar reductions in crop yield to those associated with waterlogging (Section 1.6.3). Although these reductions are a third of those caused by waterlogging, crops in Southern England are under drought stress eight years out of ten (Beran and Charnley, 1987). This is especially the case in eastern England, where the potential transpiration of corn often exceeds the summer rainfall (Briggs, 1978). Cannell *et al.*(1984) for example, have reported reductions of 7% and 9% in the yield of winter barley and winter wheat respectively due to drought in a clay soil, highlighting the need for water levels suitable for crop irrigation. An over-riding objective of ditch water level management in arable areas should therefore be that the ditch does not run dry in summer (Bill Gower, Farmer, Pers. Comm.). This statement provides a useful indication of the minimum tolerable water level required by arable farming during summer. A target water level associated with the provision of at least 0.30m of water in the ditch was adopted to satisfy the need for irrigation of the crop. In the ditch system on the SWT Reserve, this was equivalent to a water level of 1.50 m OD (0.72 m BMFL), a value adopted during the majority of the crop growth period (April to July). Due to the correspondence of the winter value of $Max_{Level\ Arable}$ and the winter water level proposed by Ritzema (1994) (1.0 m BMFL), the summer water level of 0.5 m BMFL proposed by that author was adopted as Max_{Level} during the equivalent period.

High water levels may also cause waterlogging by blocking field drains, a further consideration when developing estimates of $Max_{Level\ Arable}$. Under-drains have been installed on many areas of the Pevensey Levels (Section 2.2.2) and the effects of revised water level management prescriptions on their functioning has been one of the issues raised by local farmers when commenting on the impacts of restoration strategies applied on the wetland (Table 2.14). In under-drained areas it will be necessary to maintain ditch water levels below the drain level to ensure the successful functioning of the field drainage system (Smedema and Rycroft, 1983; Figure 6.4). For effective drainage, water levels in the channels should not cover the invert level of the land drains in the catchment. A 0.15 m freeboard should also be allowed (Beran, 1987). Based on a design depth of under-drains of 0.75m below the ground surface (Morton, 1990), the calculated value of Max_{Level} for arable, under-drained areas ($Max_{Level\ Underdrainage}$) was 0.90 m BMFL. This enabled the use of a single set of threshold values in subsequent simulations to illustrate the water level requirements of all arable practices on the wetland.

6.3.4.2. Grazing

The main issues addressed by water level management strategies in grazed areas are similar to those in arable areas, and can be summarised as:

- Limiting winter disease in stock;
- Limiting spring waterlogging that will cause reduction in grass productivity;
- Reducing poaching damage during times of waterlogging;
- Providing a sufficiently long grazing season for maximum productivity;
- Ensuring that sufficient water for irrigation of grass crop is available at times of high evaporative demand;
- Providing wet fencing;
- Ensuring gateways are not flooded.

Providing sufficient capacity in the ditch system to limit inundation and waterlogging can satisfy most of these objectives.

With grassland the effects of poor field drainage are perhaps not so evident as in arable areas (Beran and Charnley, 1987), although many agricultural grasses are relatively intolerant of high water-tables and waterlogging in spring may result in a total grass kill (RSPB *et al.*, 1997). The liver fluke, a common parasite affecting both cattle and sheep, at one stage in its lifecycle is reliant on a snail found only in damp grassland that will die if drainage is effective (Beran and Charnley, 1987). As in arable areas, the passage of agricultural plant over grassland soils when wet can have important effects on yield. The effects however are not related to changes in soil physical properties, which by their influence on soil aeration and moisture capacity are commonly quoted as important factors (Oskoui, 1992), but are related to trampling of the soil surface by depastured stock, termed poaching. On the SWT Reserve, numerous low-lying areas have been severely damaged by this process (Plate 4.3.f) leading to a reduction in the grazeable area.

The flooding of gateways has been one of the main complaints of signatories to hydrologically-based restoration strategies on the wetland (Table 2.14). Gateway submergence reduces stock mobility, so that animals are subjected to the cold and disease for longer periods (Joe Norris, Farmer, Pers. Comm.). This has been an especially important issue on the SWT Reserve, where sheep graze the nature reserve in the autumn and winter, and cattle are depastured in spring and summer. During summer, stock mobility is a similarly important issue. The lack of hedges in the typical wet grassland landscape means that ditches are commonly used as 'wet fences' to limit stock movement from field to field. In this way it is also possible to provide irrigation of the grass crop. Maps prepared by Pearl *et al.* (1954) for grassland indicate a calculated frequency of irrigation need exceeding five years in 10 for eastern England, and up to nine years in 10 in the extreme south-east (Spedding and Diekmahns, 1972).

Since both the operations of arable and pastoral farming share the objective of limiting waterlogging, values of Max_{Level} for areas of arable production (see Section 6.3.2.1) could have been employed. However, an important difference between grazed and arable land and their management is the rainfall return period employed for land drainage design in grazed areas. Drainage systems in grazed areas should be able to cope with the local 1 in 5 year rainfall event (Shaw, 1993; Table 1.11), as opposed to a 1 in 10 year event in arable areas. This difference is related to the 'value' of the comparative value of the two crops. An identical approach to that employed for the calculation of Max_{Level} in arable areas was therefore applied to determine the water level required to absorb the design storm without causing inundation of field surfaces. The 2-day M5 for the Pevensey Levels was 49mm. Based on this analysis the value of $Max_{Level\ Grazing}$ was established as 1.71 m OD, or 0.51m BMFL. This value was closely coincident for water levels proposed by Spoor and Gowing (1995) as suitable for grazing during the summer months (0.45m BMFL). The suitability of the calculated value of $Max_{Level\ Grazing}$ was further emphasised by the fact that at this water level, stock mobility and field access was guaranteed at all times. Gateways are commonly low points in the ditch catchment (Douglas 1993), and on the SWT Reserve their elevation was closely coincident with the minimum field level (Section 6.4.3). Due to the similarity between calculated values of summer $Max_{Level\ Grazing}$ and the ideal water levels proposed by Spoor and Gowing (1995), the ideal winter water level of 0.8m BMFL proposed by these authors could be adopted as the value of $Max_{Level\ Grazing}$ during the winter months.

The values of Min_{Level} calculated for grazed areas ($Min_{Level\ Grazing}$) throughout the year accounted mainly for the need to provide irrigation and wet fencing during summer. Indeed, there is direct evidence of the impacts on grass productivity on the SWT Reserve of low water levels. During the summers of 1995 and 1996 ditch water levels receded close to the dry level. During this period summer grass production on the reserve was so low that feed had to be brought in to supplement the diet of stock being grazed there by local farmers under a management agreement with the SWT (Neil Fletcher, SWT Warden, Pers. Comm.). A value for $Min_{Level\ Grazing}$ of 0.7 m below field level was adopted for grazing during the spring and summer. This was based on the ditch water level identified by the Reserve Warden as the minimum required to sustain the grass crop, and provided a ditch water depth of at least 0.4m that is appropriate to maintain wet fences (Joe Norris, Farmer, Pers. Comm.). The summer value of $Min_{Level\ Grazing}$ was adopted between April and October, which is the growth period of most grasses (NEDO, 1974). At other times of year Min_{Level} was set 0.25 m lower (0.95m BMFL). This difference is traditionally used in the UK to determine summer and winter drainage standards (Mann and Green, 1978) and coincided closely with the mean difference between summer and winter electrode settings at pumping stations on the Pevensey Levels (Table 6.9).

Sub-catchment	Mean Field Level (m OD)	Summer Electrode (m OD)	Winter Electrode (m OD)	Difference (m)
Glynleigh	2.00	+0.60	+0.40	0.20
Horseye	2.00	+0.00	-0.15	0.40
Manxey	1.40	+0.60	+0.20	0.15
Whelpley	3.50	+0.60	+0.13	0.47
Waterlot	2.00	+0.30	+0.30	0
Barnhorn	1.50	+0.25	-0.03	0.28
Star Inn	1.75	+0.60	+0.25	0.35
MEAN	2.02	+0.42	+0.15	0.26

Table 6.9. Operational electrode levels for pumping stations on the Pevensey Levels wetland, illustrating the seasonal difference between target water levels. Based on information provided by Blackmore (1993) reproduced in Table 2.7.

6.4. Evaluating the impacts of water level management on stakeholders

Summary tables of the water level preference data developed in Section 6.3 for agricultural and nature conservation stakeholders on the Pevensey Levels are given in Tables 6.10 and 6.11 respectively. In this section, those data are applied within the PINHEAD modelling system to address some of the key issues associated with historical and future water level management strategies on the wetland. The specific issues addressed have been identified from discussions of the Pevensey Levels Study Group, the operation and membership of which has been previously described in Section 2.8.3. The nature of the Pevensey Levels ensure that the majority of these issues have a strongly hydrological focus, and can therefore be addressed using the hydrological model described in Chapter 5, coupled to the data established in Section 6.3. In almost all cases the issues of importance to stakeholders on the Pevensey Levels wetland are equivalent to those apparent within the wet grassland management debate at the national scale, so that the results presented are significant in a UK context.

Members of the Pevensey Levels study group have frequently asked for best scientific opinion (Gasca Tucker and Acreman, 1999), and the application of the model to address their concerns is seen as a way of furthering scientific participation in the group. In some instances, the water level preference data developed in Section 6.3 have been incorporated within the model as sluice level regimes. By setting these sluice levels within the model, and applying other water level preference data within the PINHEAD_Hydroecology Module, it is possible to examine the correspondence between, or quantify the impacts associated with, the water level requirements of two distinct stakeholders groups. The issues addressed within the modelling framework are mainly those identified in Table 2.14 which highlights some of the recurring themes associated with the management of the Pevensey Levels. The simulations are complemented by addressing other issues of interest described in Chapter 2, including the causes of the decline of the ornithological value of the site and helping to explain the distribution of key species of nature conservation importance on the wetland. In doing so, the effects of higher water levels on a variety of wetland hydrological processes is also provided, extending previous catchment and field-based analyses of the wetland water balance presented in Chapters 3 to 5.

Management objective	Water Level (m BMFL)	Period of importance
Arable farming		
• Limiting waterlogging will result in reductions in crop productivity and land workability	Max_{level} 0.87 Max_{level} 0.50	*September – March* *April - August*
• Maintaining water levels below field drains	Max_{level} 0.90	*All year*
• Maintaining water levels for crop irrigation in summer	Min_{level} 0.82	*April - August*
Grazing		
• Limiting waterlogging that will result in poaching and stock disease	Max_{level} 0.51 Max_{level} 0.80	*April-October* *November-March*
• Limiting waterlogging that will result in reductions in grass productivity	Max_{level} 0.80	*November-March*
• Prevention of gateway flooding (ensuring cattle mobility)	Max_{level} 0.22	*All year*
• Maintaining water levels for crop irrigation and wet fencing in summer	Min_{level} 0.7	*April-October*

Table 6.10. A review of the water level requirements of agriculture. Based on data established in Sections 6.3.2.1 and 6.3.2.2.

Species type	Species name	Common name	Max_{Level} (m BMFL)	Min_{Level} (m BMFL)	Period of importance	National status	Status on the Pevensey Levels
Flora	*Potomageton acutifolius*	Sharp leaved	0.32	0.92	April-September	Nationally Rare	Common
	Potomageton trichoides	Pondweed	0.12	0.92	April-September	Nationally Scarce	Common
	Stratiotes aloides	Hairlike Pondweed	0.12	0.92	April-September	Nationally Scarce	Common
	Sium latifolium	Water soldier	0.72	1.42^{+}	April-September	Nationally Scarce	Common
	Urtticularia sp.	Greater water parsnip	0.12	1.22^{+}	April-September	Nationally Scarce	Common
	Hottonia palustris	Bladderwort	0.32	1.17^{+}	April-September	Nationally Scarce	Common
		Water violet					
Arachnid	*Dolomedes plantarius*	Fen Raft Spider	0.00	0.80	All year	Nationally Rare	Common
Birds	*Vanellus vanellus*	Lapwing	0.28*	0.08*	November – March	Declining	Scarce
	Gallinago gallinago	Snipe	0.28*	0.03*	April	Declining	Scarce
	Tringa totanus	Redshank	0.10	0.45	May-June	Declining	Scarce
	Anatidae sp.	Anatids	0.45	0.82	August – October	Declining	Scarce

*indicates water level requirements above the mean field level. $^{+}$Indicates tolerance of water levels below the root level (plants only).

Table 6.11. Water level requirements of species of nature conservation importance on the Pevensey Levels wetland. Data shown summarise the findings of Section 6.3.1 and are used to evaluate the effects of various sluice management regimes on the nature conservation value of the wetland.

6.4.1. EXPLAINING ORNITHOLOGICAL DECLINE

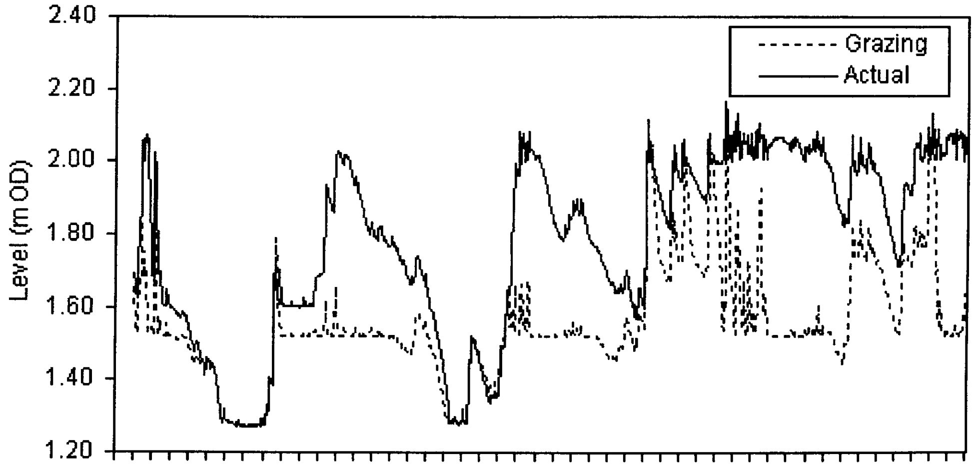
One of the main points of concern highlighted by conservationists on the wetland has been the progressive decline in the numbers of breeding and over-wintering waders and anatids on the wetland. This decline has been well documented both on the Pevensey Levels (Hitchings, 1987; Section 2.7) and on other wet grasslands in the UK (Figure 1.3). For the Somerset Levels, Green and Robins (1992) have ascribed the reduction in bird numbers to decreasing pump start levels (Section 1.2.3). Lower pump start levels reduce the frequency and duration of flooding in line with one of the main objectives of water level management for agriculture (Sections 6.3.4). However, a similar study, relating pump start levels and bird numbers (Figure 1.4) was not possible on the Pevensey Levels due to the lack of historical pump start data. As a result, PINHEAD was employed to quantify the impacts of pump-drainage and water level management for agriculture on the characteristic bird species of wet grassland (Section 1.2.3).

Information describing the water level preferences of wet grassland bird species developed in Section 6.3.3 were employed to quantify the specific impacts associated with past and current water level management for agriculture. In terms of areal extent, agriculture on the Pevensey Levels wetland has historically been dominated by grazing (Section 2.2.2). As a result, the main focus of the assessment is the impact of water level management for grazing on habitat suitability for birds. This is achieved by applying sluice levels accordant with the water level requirements of grazing throughout the year. Based on Section 6.3, target water levels associated with grazing can be broadly summarised as 0.5 m below field level between April and September and 0.7 m below field level at other times (Spoor and Gowing, 1993; Table 6.10). This sluice management regime has been incorporated as one of the options within the PINHEAD_Sluice_Levels_Input Module. This sluice level regime, labelled 'Agriculture (Grazing)', can be selected in the 'Water Level Management Options' frame of the PINHEAD_Options Module (see Box 6.2).

Ditch water levels on the SWT Reserve simulated by application of a sluice management regime suitable for grazing are shown in Figure 6.5.a relative to actual water levels during the equivalent period. Also shown are the effects of the water level management regime for grazing on habitat suitability for birds (Figure 6.5.b). The precise values of Max_{Level} and Min_{Level} adopted for birds at different times of year are those shown in Figure 6.2. Implementation of a sluice management regime for agriculture results in large negative exceedences throughout the entire four-year period. Negative exceedences indicate water levels lower than those required by the target species. The largest negative exceedences are associated with the winter period, when the main objective in farmed areas is to evacuate excess flood water to limit waterlogging and inundation (Section 6.3.2). This is also the time of year when large inundated areas are required by avian species (Section 6.3.3). By setting sluice levels in accordance with the requirements of grazing, at no time between 1995 and 1998 do water levels approach the mean field level, and only for 8 days during 1997 do water levels exceed the minimum field level (Figure 6.5b). The maximum simulated water level between 1995 and 1998 was 2.07m OD on the 28th June 1997, associated with an intense storm that generated in excess of 50 mm of rainfall in the preceding five days.

Large negative exceedences during the summer months of 1995 and 1996 contrast with smaller exceedences recorded during the summers on 1997 and 1998 (Figure 6.5.b). Inter-annual differences can be explained by the prevailing climatic conditions in each of these years. Water level records for the SWT Reserve have shown that ditches were effectively dry during the summers of both 1995 and 1996 (Section 3.6.1). The model predicts that the implementation of water levels suitable for grazing actually leads to an increase in the duration of these episodes during dry years. Model predictions based on actual sluice settings estimate that ditches were dry for 117 days and 63 days during 1995 and 1996 respectively (Table 6.12). Implementation of a sluice management for grazing increases the duration of these events by 5 and 14 days in 1995 and 1996 respectively relative to actual settings (Table 6.12). Reverting to summer settings before the traditionally employed month of April, a common practice on the wetland in dry years (M. Harding, Grazier, Pers. Comm.), fails to cause a substantial reduction in the duration of 'dry' conditions. Model results for 1995 and 1996 suggest that the frequency of water levels less than 1.40 m OD would be reduced by only 5 days by reverting to summer sluice levels at the end of March and by 11 days by reverting to summer levels at the end February (Table 6.12). This latter reduction is equivalent to that recorded when sluices are maintained at summer levels all year round (Table 6.12), highlighting the difficulty of limiting the impacts of dry summers on the Pevensey Levels wetland by providing winter storage, an issue also discussed in Section 6.4.4.

(a)



(b)

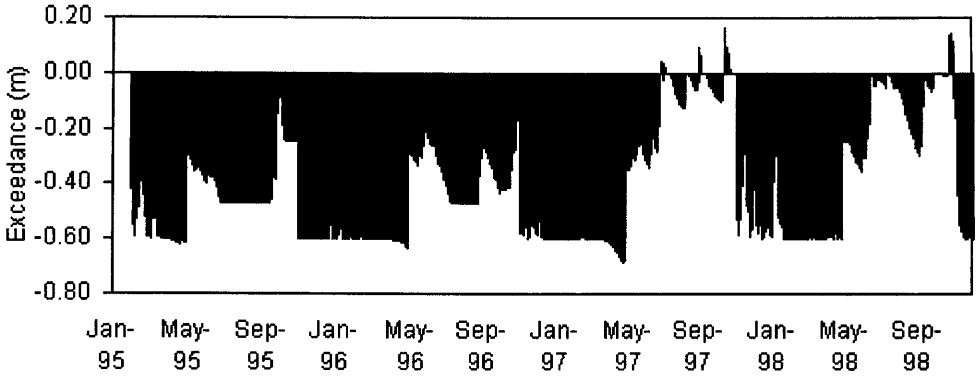


Figure 6.5. (a) Actual water levels on the SWT Reserve 1995-1998 relative to those resulting from the implementation of a sluice management regime for grazing. (b) shows the exceedences associated with implementing sluice levels for grazing on birds.

Sluice management option	1995	1996	1997	1998
Actual	117	63	0	1
Grazing (summer settings in April)	122	77	0	1
Grazing (summer settings in March)	117	72	0	1
Grazing (summer settings in February)	111	66	0	1
Grazing (summer settings all year round)	111	66	0	1

Table 6.12. Annual frequency (in days) of water levels less than 1.40m OD of adopting different sluice level management regimes on the SWT Reserve.

6.4.2. EXPLAINING CURRENT BIODIVERSITY VALUE

Although for most wet grassland areas in the UK birds remain the key target species, on the Pevensey Levels biodiversity value is based strongly on the flora and fauna inhabiting the ditches (Section 2.6). Previous surveys of both rare and scarce flora (Keymer *et al.*, 1989) and the fen raft spider, *Dolomedes plantarius* (Jones, 1992), have found these species to be concentrated mainly in the gravity drained area of the wetland (Figures 2.14 and 2.16). Indeed, in only a very few cases are any individuals of these species found in arable or pump-drained areas (Glading, 1986). Differences between the hydrology of pump- and gravity-drained areas of the wetland, especially with respect to ditch water levels (see Chapter 3), indicate that there is indeed a hydrological case to answer within any description of the causes of biodiversity decline on the site.

The observed distribution of rare and scarce flora on the wetland and that of *Dolomedes plantarius* has been investigated by comparing the exceedences beyond the requirements of these species of adopting sluice management regimes for grazing or maintaining water levels in accordance with those recorded at pumping stations. For the purpose of simplicity, in the simulation described, *Dolomedes plantarius* and rare floral species (*Potomageton acutifolius*, *Potomageton trichoides*, *Stratiotes aloides*) have been modelled as a community since their water level requirements are broadly similar (Table 6.11). Water level preference data input to the PINHEAD_Hydroecology Module for this community-based simulation is summarised in Table 6.13.

The water levels resulting from the application of sluice management regime for grazing on the SWT Reserve have been previously shown in Figure 6.5. In Figure 6.6a they are shown alongside water levels resulting from the implementation of a sluice level regime that simulates the control influenced by pumping on ditch water levels. The sluice management regime employed to simulate a pump is shown in Figure 6.6b, and has been derived from the ideal water levels for pump-drained areas for arable agriculture shown in Figure 1.11. Water levels maintained in pump-drained channels of the Pevensey Levels wetland (Figure 3.23) have been previously shown to be closely related to these target water levels (Section 3.5.1).

Species	Min_{Level}	Max_{Level}	$Month_{First}$	$Month_{Last}$
Dolomedes plantarius	0.00	0.80	1	12
Stratiotes aloides	0.12	0.92	4	9
Potomageton acutifolius	0.32	0.92	4	9
Potomageton trichoides	0.12	0.92	4	9

Table 6.13. Input water level preference data (m below mean field level) used for the simulation of the impacts of grazing and pump-drainage on the ditch flora and fauna of the Pevensey Levels wetland.

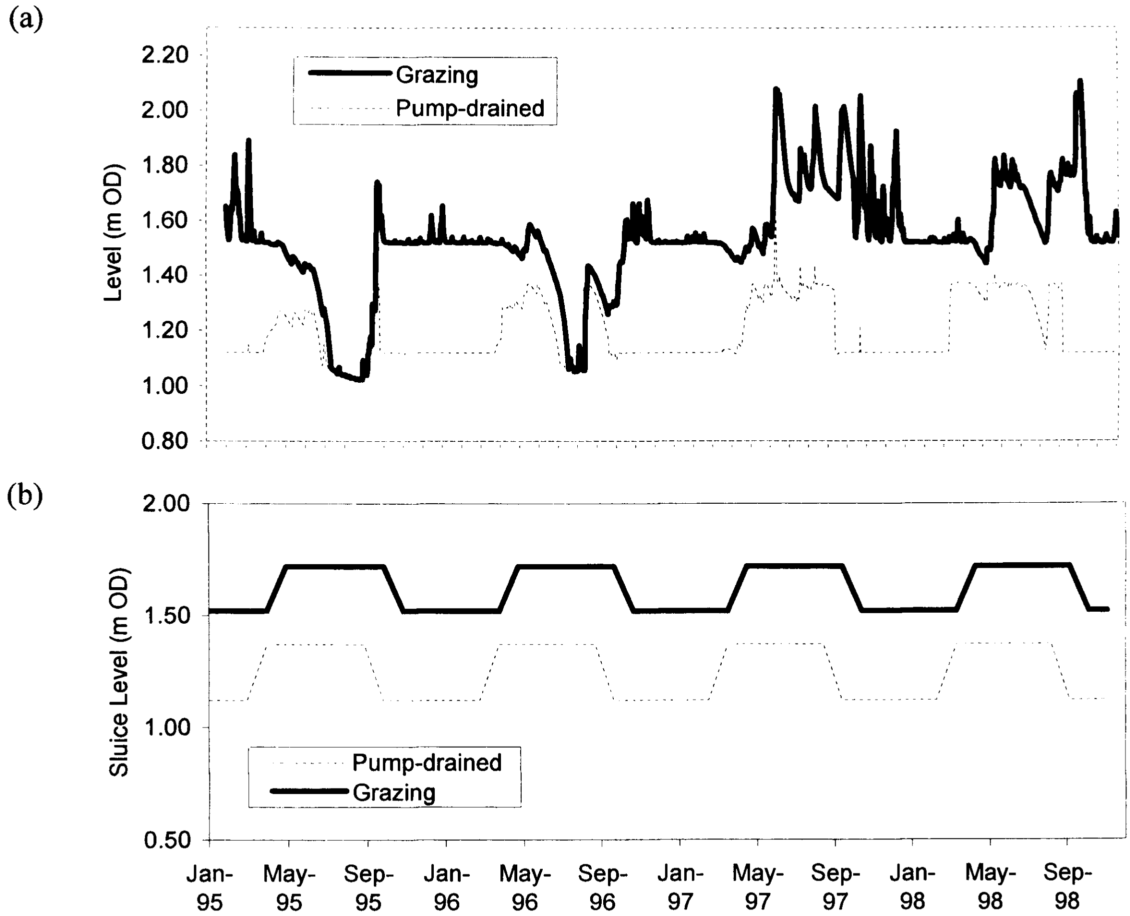


Figure 6.6. (a) Water levels on the SWT Reserve simulated using sluice levels associated with grazing and pump-drainage. (b) shows the specific sluice level regimes employed.

Model results indicate that the sluice management regimes commonly adopted for grazing are highly suited to the requirements of ditch flora and fauna. Limited exceedences beyond the water level requirements of ditch flora and fauna are recorded throughout the entire four-year period based on the implementation of a sluice level regime for grazing. The largest exceedences are associated with the summers of 1995 and 1996 when water levels dropped below species requirements for 91 days and 97 days respectively. The largest SEVs were recorded during the summer of 1995 (Figure 6.7a). This was associated with a continuous period during which ditches were dry. Ditches were also dry for a considerable proportion of the summer of 1996. SEVs recorded during 1996 were however lower than those during the summer of 1995 since water levels rose during August 1996 following a dry period in June and July, although by September water levels had again receded beyond species requirements. It is assumed that exceedences such as those in 1996, characterised by a number of exceedences of short duration will have less impact on species than single, long duration episodes such as that during the summer of 1995. For 12 days during the summer of 1997, water levels were also slightly higher than the requirements of the ditch community, although this event recorded a small Sum Exceedence Value (SEV) (Figure 6.8a).

In terms of their duration however, the exceedences associated with water level management for grazing were a fraction of those recorded by the model when the effects of pumping on the SWT Reserve were simulated (Figure 6.7b). Negative exceedences were recorded on every day of the four-year study period. The limited magnitude of the exceedences is a feature of the water level preference data. Indeed, pumping from ditches on the SWT Reserve based on the mean electrode levels apparent on the wetland causes the ditches to remain dry between October and April in all years (see Figure 6.6a). This is expressed as the flat aspect of the base of the exceedence chart shown in Figure 6.7b. The large exceedences recorded in the model simulation highlight the potentially large impacts of pumping on the flora and fauna of the Pevensey Levels wetland. The results also provide a clear means of explaining why the rare species characteristic of wetland ditches are concentrated in gravity-drained areas of the Pevensey Levels, outside the area of influence of the pumps (see Section 2.6 and 2.7).

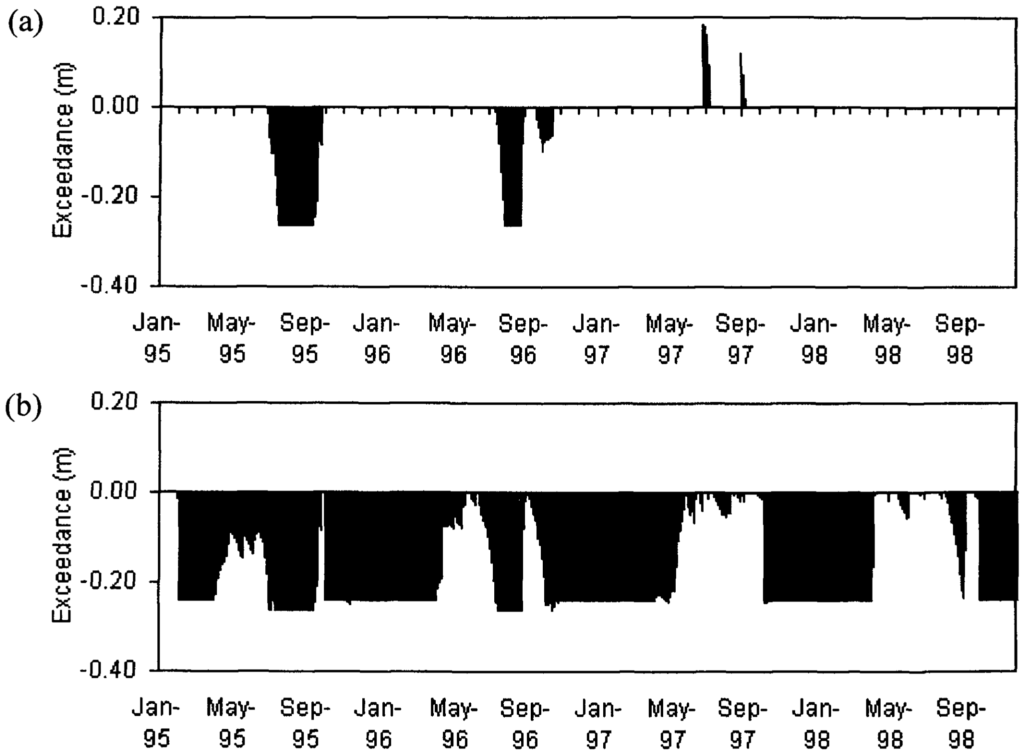


Figure 6.7. Impacts on *Dolomedes plantarius* and rare/scarce flora of sluice level regimes for (a) grazing and (b) pumping.

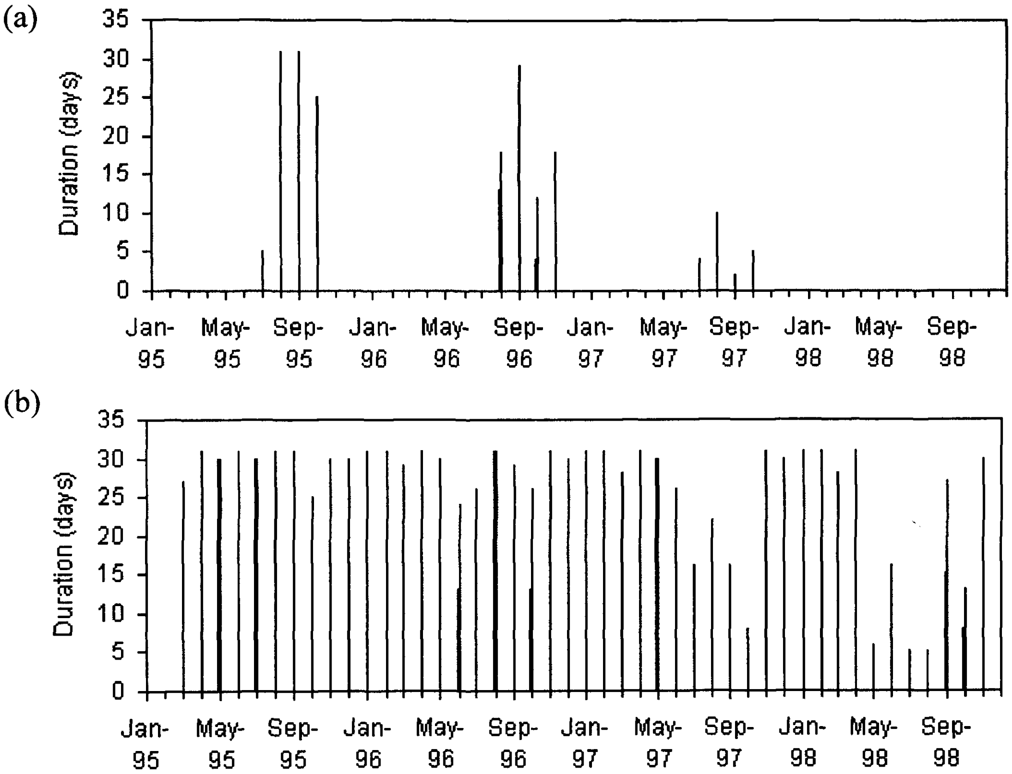
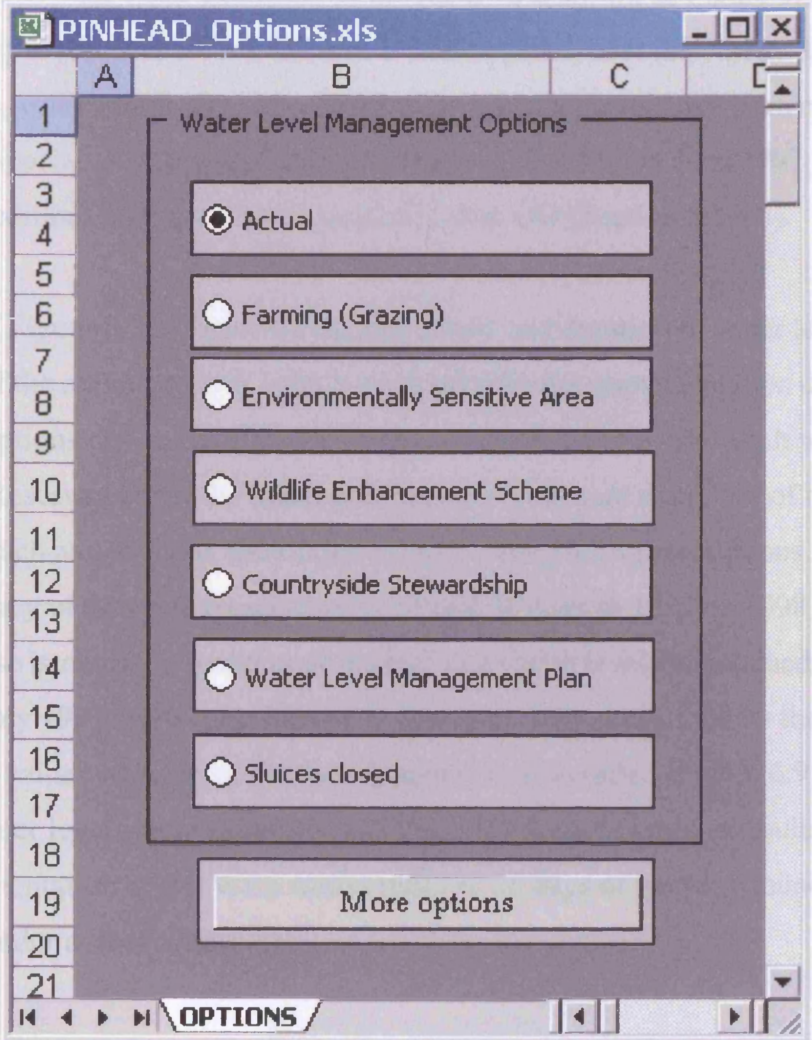


Figure 6.8. Sum Exceedence Values beyond the requirements of rare ditch flora and fauna associated with water level management for (a) grazing and (b) pumping.

6.4.3. THE WILDLIFE ENHANCEMENT SCHEME

The Wildlife Enhancement Scheme (WES) has been the main tool employed to restore the biodiversity value on the Pevensey Levels wetland. Section 2.8.1 has provided a general review of the issues associated with the implementation of the scheme on the Pevensey Levels and the prescriptions associated with the scheme have been reviewed in Table 2.13. In general, the WES can be considered exemplary in the way in which water level management strategies can be revised for ecological benefit with the consensus of the farming community. The large uptake of the scheme has been generally related to the fact that WES prescriptions do not provide a large variation from current management. An important component has been that by not advocating winter flooding, the scheme, in principle at least, has the wide support of the local farming community. Indeed, for many farmers who manage their land along traditional lines, the WES is simply formalising what they already do (Whitbread and Curson, 1992).

However, the continuation of the WES is reliant on solving some of the problems that have been associated with the scheme to date. These problems are mainly related to the water level management prescriptions (Table 2.14). For example, in seeking water level management regimes that satisfy all wetland stakeholders, the WES has been criticised by conservationists for not promoting surface inundation required by wet grassland birds. From the farmers perspective, an important effect of the scheme has been the flooding of gateways that limit access to the land. In arable areas, farmers have reported that WES water levels submerge under-drains, with consequent effects on crop yields (Bill Gower, Landowner, Pers. Comm.). These problems highlight the difficulty of providing an integrated ditch water level management regime to satisfy the multi-sectoral nature of management on the Pevensey Levels wetland. In this section, the main problems associated with the WES have been addressed by application of the PINHEAD model. To examine the impacts on farming and the benefits accrued by wetland biota it has been assumed that to attain WES target water levels, sluices will be maintained at levels equivalent to the prescribed water levels given in Table 2.13: 0.30m below mean ground level (BMFL) between January and August and 0.60m BMFL at other times. This scheme has been incorporated in the PINHEAD_Sluice_Levels_Input Module (Box 5.8) and can be selected in the 'Water Level Management' frame of the PINHEAD_Options Module (Box 6.2).



Box 6.2. PINHEAD_Options Module in showing the sluice settings associated with different water level management options that can be run within the model.

6.4.3.1. WES and the inundation of gateways

Ditch water levels resulting from the implementation of the WES sluice levels on the SWT Reserve are shown in Figure 6.9 alongside water levels simulated using actual sluice settings between 1995 and 1998. To evaluate the influence of these sluice settings on the inundation of gateways, gateways have been levelled to the same datum as the Field 2 water level recorder and the mean gateway level calculated based on the elevation of the three gateways on the Reserve is shown in Figure 5.5. The mean gateway level on the SWT Reserve is 2.05m OD. Under the current sluice management regime, water levels in excess of the mean gateway level have been common (Figure 6.9), especially following the re-profiling of sluice P26 in June 1997, which has raised the maximum retainable water level to 2.09m OD (Section 5.3.4.3).

Figure 6.10a identifies the magnitude and duration of water level exceedences beyond the mean gateway level associated with the implementation of WES water level prescriptions on the SWT Reserve. Using actual sluice levels, ditch water levels on the SWT Reserve exceed the mean gateway level for more than 20% of the year in wetter years such as 1997 and 1998 (Table 6.14). Using WES prescriptions reduces the frequency of gateway inundation by 63 and 50 days in 1997 ad 1998 respectively. This decrease is mainly a function of the fact that sluice levels maintained on the Reserve after July 1997 were considerably higher than those prescribed by the WES, especially during winter when most gateway inundation is recorded (Figure 6.9). In 1995, when low water levels were maintained on the SWT Reserve, model results suggest implementation of the WES would result in 35 days of gateway inundation relative to none under actual conditions.

The effects of other water level management options on gateway inundation frequency are summarised in Table 6.14. Implementation of a sluice level management regime for grazing or arable farming avoids gateway inundation for the majority of the four-year period (Table 6.14). In contrast, for schemes that primarily satisfy nature conservation objectives (ESA, Countryside Stewardship), the model predicts large increases in the frequency of gateway inundation. In combination, model results highlight the need to re-profile gateways prior to scheme implementation, an issue previously noted by the local farming community (Table 2.14). For all schemes, indicative gateway levels required on the SWT Reserve are given in Table 6.14.

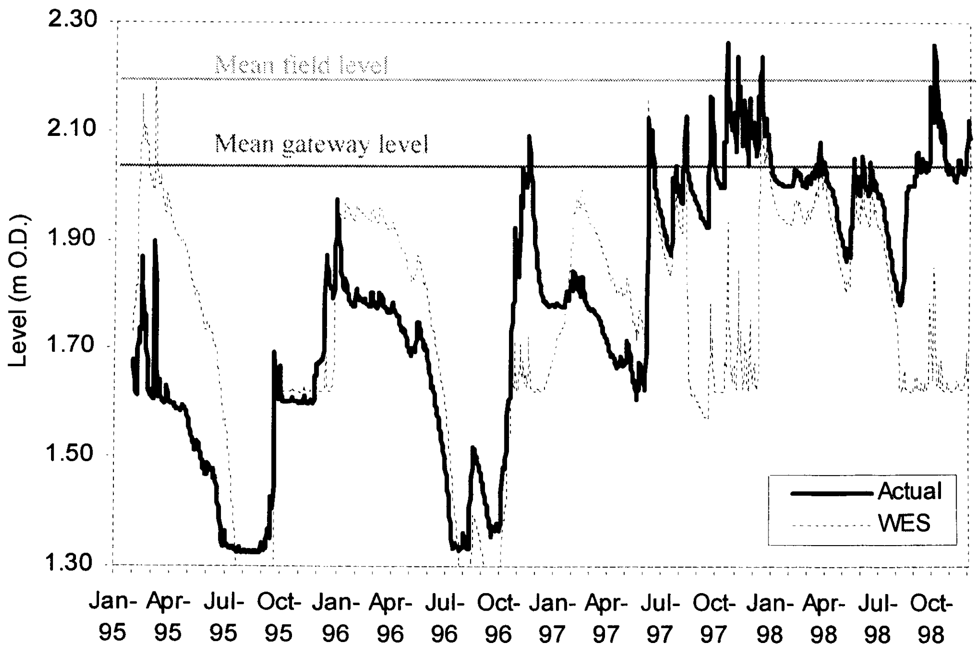
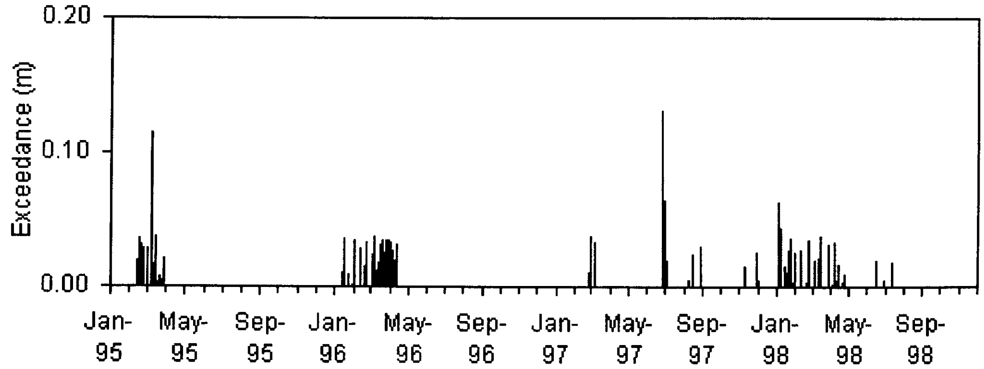


Figure 6.9. Ditch water levels resulting from a sluice level management regime coincident with WES water level prescriptions on the SWT Reserve.

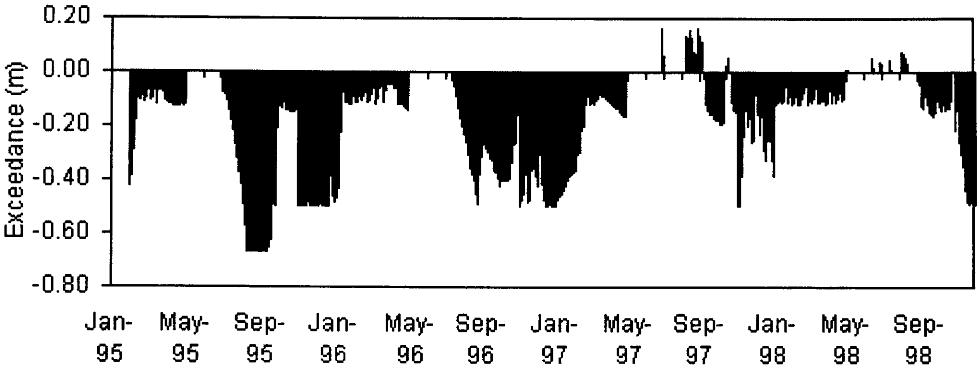
Sluice management regime	1995 (%)	1996 (%)	1997 (%)	1998 (%)	Gateway level to ensure no submergence (m OD)
Actual	0.0	1.4	21.4	22.8	2.26
WES	9.6	0.0	4.1	9.1	2.23
Arable	0.0	0.0	0.0	0.0	1.48
Grazing	0.0	0.0	1.1	0.0	2.07
Countryside Stewardship	10.5	8.5	22.7	23.6	2.25
ESA	10.5	8.5	22.7	23.6	2.27

Table 6.14. Frequency of inundation of gateways on the SWT Reserve (% of the total year) under different water level management strategies. In each case, the gateway level required to ensure no submergence is also given. The mean actual gateway level on the SWT Reserve is 2.02m OD.

(a)



(b)



(c)

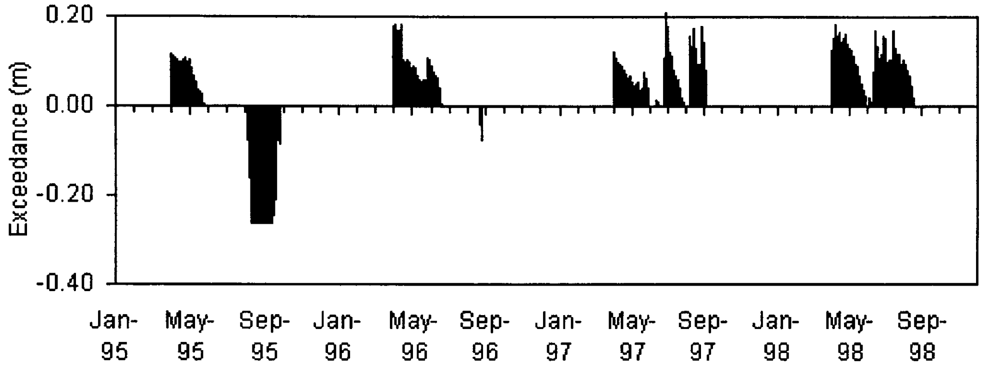


Figure 6.10. Exceedences beyond (a) the mean gateway level, (b) the water level requirements of birds and (c) the requirements of ditch flora and fauna due to implementation of WES water level prescriptions on the SWT Reserve. Note the different y-axis scales.

6.4.3.2. WES and species of conservation importance

The PINHEAD_Hydroecology Module has also been employed to address the concerns frequently highlighted by wetland stakeholders with regards to the suitability of WES to birds. Figure 6.10.b shows the exceedences beyond the requirements of wet grassland birds associated with the implementation of WES water level prescriptions on the SWT Reserve. Results illustrate that although the inundation of gateways is a frequent event under WES water level prescriptions (Figure 6.9), episodes of field inundation are of insufficient frequency and magnitude to provide suitable conditions for the characteristic bird species of wet grassland. Difficulties are compounded by the large differences in the areal extent of flooding on the SWT Reserve that result from limited changes in water levels. Only 0.1% of the total SWT Reserve area is inundated at 2.00m compared with 10% at 2.14m OD and 29% at 2.22m OD, the mean field level on the SWT Reserve (Figure 6.1).

Although water levels in excess of 2.00m OD, the minimum field level, are recorded during 15.3%, 19.7%, 15.9% and 41.9% of each year during the period 1995-1998, implementation of a sluice level management regime accordant with WES results in the mean field level being exceeded on only three occasions during the entire four year period. As a result, large, almost continuous negative exceedences are a feature of the relationship between the water level requirements of birds and the water level associated with the WES (Figure 6.10b). The largest exceedences are recorded during the winter months (Figure 6.10b), when the lower WES water level prescriptions adopted to satisfy the requirements of farming on the wetland coincide with the period of maximum inundation required by bird species (Section 6.3.3). In contrast, the water levels prescriptions associated with the WES are highly suitable for the rare flora and fauna community inhabiting the ditches of the Pevensey Levels. Only in July 1997, when model predictions suggest a combination of WES summer sluice settings and intense summer rainfall would lead to a brief period of field inundation, do water levels exceed thresholds developed for the ditch flora and fauna community. This results in visibly smaller exceedences during the four year period when compared to either birds or gateway inundation (Figure 6.10.c).

Year	Exceedence statistic	Ditch flora and fauna	Birds	Gateway inundation	Grazing	Arable cropping
1995	Total exceedence (m)	-0.40	-71.43	+3.42	+65.54	+80.39
	Total duration (days)	44	261	44	271	272
	Mean exceedence (md^{-1})	0.00	-0.27	+0.07	+0.18	+0.30
1996	Total exceedence (m)	0.00	-74.40	+0.02	+88.08	+95.33
	Total duration (days)	0	292	6	288	293
	Mean exceedence (md^{-1})	0.00	-0.25	0.00	+0.24	+0.33
1997	Total exceedence (m)	1.74	-54.95	+2.01	+100.65	+120.46
	Total duration (days)	32	280	34	318	365
	Mean exceedence (md^{-1})	0.02	-0.20	+0.34	+0.28	+0.33
1998	Total exceedence (m)	0.00	-43.86	+3.34	+114.55	+135.01
	Total duration (days)	0	257	80	312	365
	Mean exceedence (md^{-1})	0.00	-0.17	+0.04	+0.31	+0.37

Table 6.15. Comparative evaluation of the impacts of WES on various stakeholders on the Pevensey Levels wetland. *Mean exceedence is calculated by Total Exceedence/Total duration.

A summary of the impacts of WES water level prescriptions on all stakeholders on the Pevensey Levels wetland is given in Table 6.15 on a year-by-year basis. This table provides a comparative overview of the exceedences beyond the water level requirements of various stakeholders associated with the implementation of WES water level prescriptions. In terms of both magnitude and duration, exceedences beyond the water level requirements of birds and arable agriculture simulated by the model are equivalent. However, WES water levels are too low for birds and too high for arable farming, as previously highlighted by a number of members of the Pevensey Levels Study Group. WES water levels were also too high for graziers, resulting in high mean daily exceedence values. However, the fact that the total and mean daily exceedences beyond the requirements of grazing are smaller than those associated with arable farming illustrate why traditional graziers on the wetland represent the main bulk of signatories to the WES (Section 2.8.1).

6.4.4. SUSTAINABILITY OF PROPOSED WATER LEVEL MANAGEMENT

Due to the difficulties associated with providing water level conditions favourable for birds based on Wildlife Enhancement Scheme (WES) water level prescriptions, a variety of alternative water level regimes have been proposed for the Pevensey Levels wetland. In 1991 a feasibility study was undertaken by East Sussex County Council to evaluate the possibility of obtaining Environmentally Sensitive Area (ESA) status for the Pevensey Levels (Section 2.6). A small proportion of the wetland is also under the Countryside Stewardship (CS) scheme (Figure 2.17), the water level prescriptions of which are also of benefit to wet grassland birds (Figure 1.15). However, any extra demand for water needs to be examined in the context of water resource availability. Section 3.7.2 has illustrated the difficulties of storing sufficient water in embanked channels for re-distribution to lowland areas as a means of satisfying revised water level management strategies. Analysis summarised in Figure 3.46 has shown that current water level management approaches, coupled with prevailing climatic conditions during most summers, are incapable of satisfying the increased water resource demand associated with the wetland-wide implementation of WES water level prescriptions.

A frequently quoted alternative to feeding water from embanked channels to lowland areas is to simply retain winter rainfall within the field-scale ditch systems. For the Pevensey Levels, this management option has been investigated by implementing sluice level regimes accordant with CS and ESA scheme water level prescriptions. Water level prescriptions associated with both schemes have been previously shown in Figure 1.15. Implementation of these water level regimes within the PINHEAD model has been undertaken to evaluate the effects of such sluice management regimes on habitat suitability for birds. The effects of retaining higher winter water levels on the frequency, duration and likelihood of ditches drying out during dry summers, such as 1995 and 1996, have also been investigated. Because water level prescriptions associated with ESA Tier 1 and 2 scheme do not represent a large variation from current management on the wetland, only Tier 3 water level prescriptions have been implemented within the model (at field level between December and April and no more than 0.3m below field level at other times). CS water level prescriptions can be broadly summarised as at field level between November and March and no more than 0.2m below field level at other times of year. Both sluice management regimes can be selected in the 'Water Level Management' frame of the PINHEAD_Options Module, previously shown in Box 6.2.

6.4.4.1. The influence of the CS and ESA schemes on ditch water levels

Simulated water levels associated with the implementation of ESA and CS water level prescriptions on the SWT Reserve are shown in Figure 6.11a alongside water levels simulated using actual sluice settings on the SWT Reserve between January 1995 and December 1998. Figure 6.11.b shows changes to actual water levels resulting from the implementation of ESA and CS sluice levels on the SWT Reserve during the same period. Model results indicate that implementing sluice regimes in accordance with ESA Tier 3 and CS prescriptions would have a large impact on water levels on the SWT Reserve. Changes in sluice settings create large increases in water levels relative to actual conditions, particularly prior to July 1997, when sluice P26 was re-profiled (Section 5.5.2). More limited changes in water levels on the Reserve are recorded for the latter half of 1997 and the whole of 1998 (Figure 6.11.a), when sluice levels on the Reserve were already maintained at levels similar to those proposed by the ESA and CS schemes.

The influence of ESA Tier 3 and CS sluice level regimes on the SWT Reserve water level duration curve between 1995 and 1998 is shown in Figure 6.12. The ESA and CS schemes result in large increases in the frequency of water levels between 1.60m OD and 2.00m OD (Figure 6.12). However, due to the similarity of the water level prescriptions associated with each scheme, limited differences are apparent when the water level duration curves for the ESA Tier 3 and CS water level prescriptions are compared (Figure 6.12). This trend is also apparent in plots of the time series for each scheme (Figure 6.11a) or when changes from actual water levels are considered (Figure 6.11b). Smaller changes to the frequency of water level 'extremes' (in excess of 2.00m OD and less than 1.40m OD) are associated with the implementation of CS or ESA water level prescriptions when considered relative to water levels predicted based on actual sluice settings. Model predictions illustrate that the influence of dry summers (1995 and 1996) on ditch water levels is not significantly reduced by maintaining higher winter ditch water levels (Figure 6.11.a and b). Figure 6.13.a shows that the frequency of water levels less than 1.40m OD during 1995 and 1996 is reduced by implementation of WES, CS and ESA Tier 3 prescriptions, but that these reductions are small. Differences between actual water levels and the implementation of the ESA scheme are equivalent to a reduction in the period when ditches were dry of only 27 days in 1995 and 17 days in 1996.

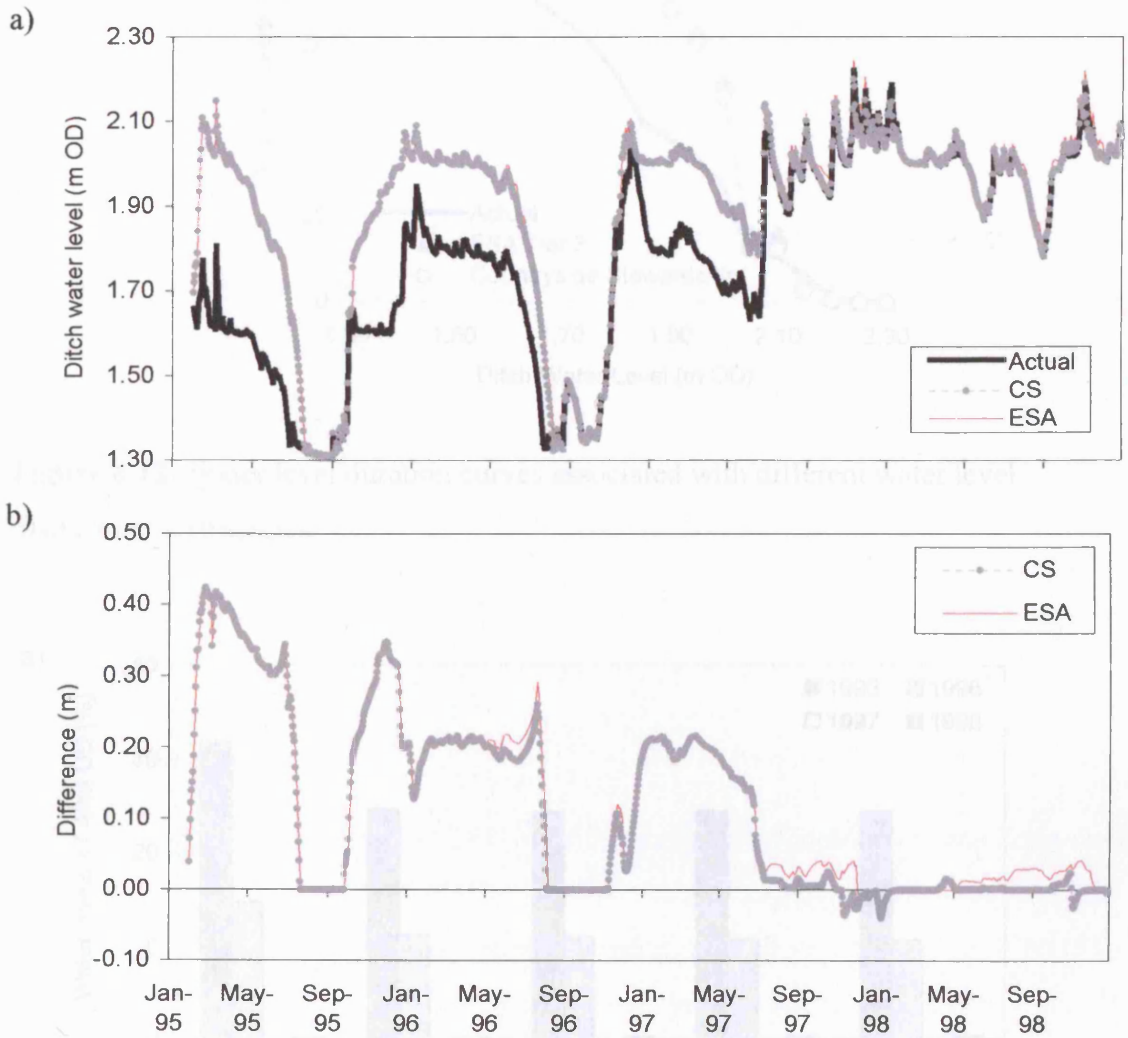


Figure 6.11. (a) Water levels predicted for ESA Tier 3 and Countryside Stewardship sluice settings on the SWT Reserve and (b) differences between simulated water levels and actual water levels on the SWT Reserve January 1995-December 1998.

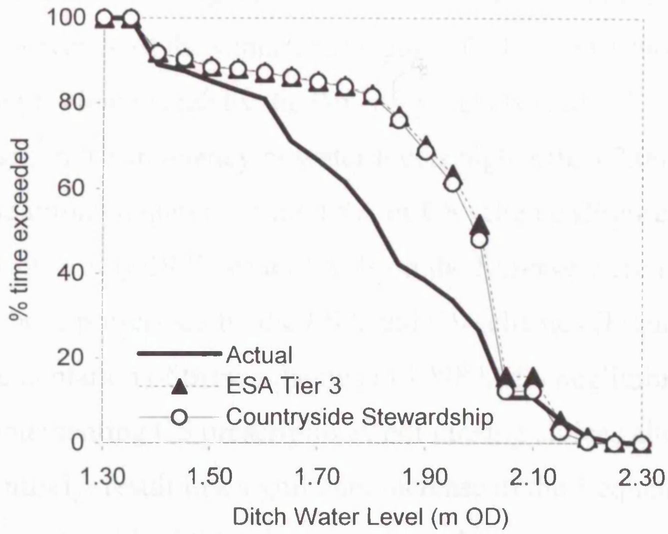


Figure 6.12. Water level duration curves associated with different water level management strategies.

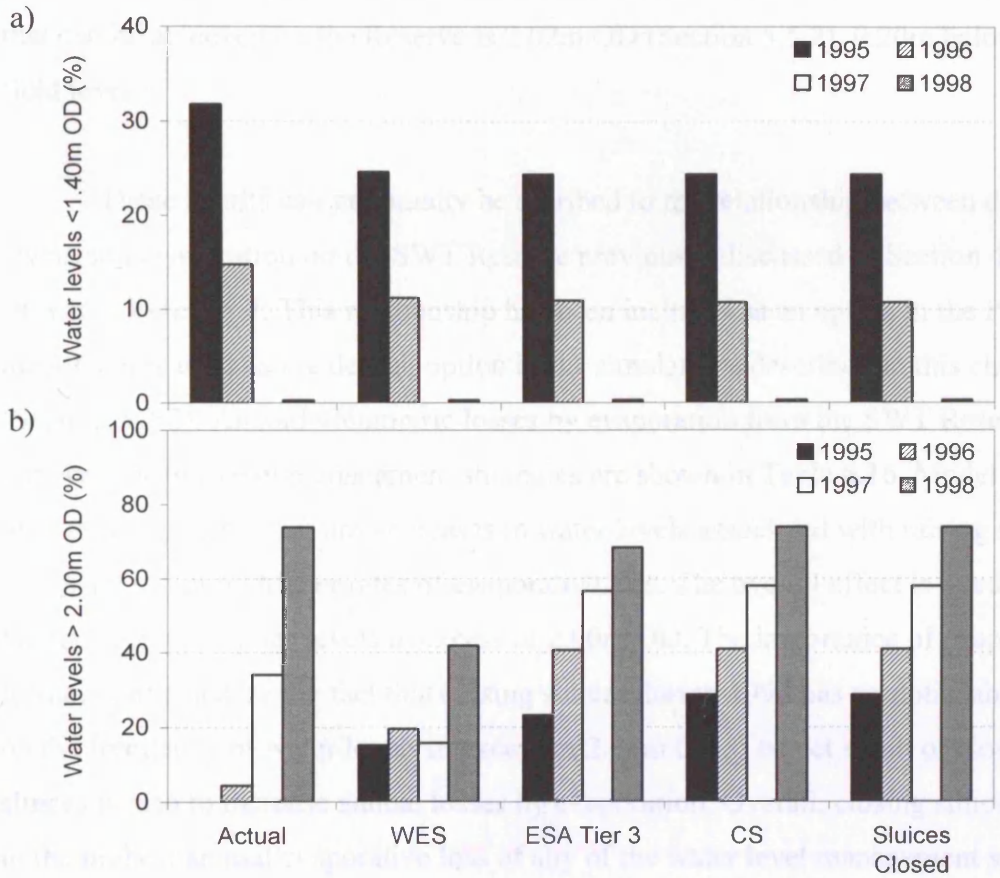


Figure 6.13. Effects of the implementation of various sluice management strategies on the frequency of water levels (a) less than 1.40m OD and (b) more than 2.00m OD.

6.4.4.2. The influence of the CS and ESA schemes on evaporative loss

The limited effects of retaining higher winter water levels on summer water level trends illustrates both the severity of the summer droughts of 1995 and 1996 and, more widely, the water resource problems faced by the Pevensey Levels wetland. In 1995, 1996 and 1997 large increases in the frequency of water levels higher than 2.00m OD are associated with the implementation of the ESA or CS scheme (Figure 6.13.b). This is mainly because prior to July 1997, water levels on the Reserve were maintained at levels lower than those prescribed by the ESA and CS schemes (Figure 1.17). In contrast, the implementation of these schemes in 1998 had a negligible impact on water levels. Neither implementing CS prescriptions, nor closing sluices altogether to isolate the ditch system entirely, result in a significant increase in the frequency of the higher water levels. In the case of the ESA scheme, raising sluice levels actually reduces the frequency of water levels greater than 2.00m OD by 20 days relative to actual sluice levels (Figure 6.13.b). This is the case even though water level prescriptions associated with both the CS and ESA schemes are in excess of the sluice levels maintained on the SWT Reserve throughout that period. As previously stated, the maximum sluice level that can be achieved on the Reserve is 2.02m OD (Section 5.5.2), 0.20m below mean field level.

These results can potentially be ascribed to the relationship between ditch water levels and evaporation on the SWT Reserve previously discussed in Section 4.7.3 and shown in Figure 4.7. This relationship has been included as an option in the PINHEAD model and is used as the default option in the simulations described in this chapter (Section 5.3.2). Annual volumetric losses by evaporation from the SWT Reserve for a variety of water level management strategies are shown in Table 6.16. Model predictions indicate that any increases in water levels associated with raising sluice levels are offset by higher rates of evaporative loss. The overall effect is a reduction in the frequency of water levels in excess of 2.00m OD. The importance of evaporation is further confirmed by the fact that closing sluices during 1998 has no noticeable effect on the frequency of water levels in excess of 2.00m OD. The net result of closing sluices is also to increase annual losses by evaporation. Overall, closing sluices results in the highest annual evaporative loss of any of the water level management scenarios considered.

Water Level Management Scheme	1995	1996	1997	1998
Actual	8,650	11,161	28,245	22,479
Countryside Stewardship	18,178	16,139	31,178	21,420
ESA Tier 3	18,172	16,139	30,211	21,055
Sluices Closed	18,178	16,697	34,639	22,180

Table 6.16. Volumetric losses by evaporation (m^3) from the SWT Reserve on an annual basis under a variety of water level management scenarios.

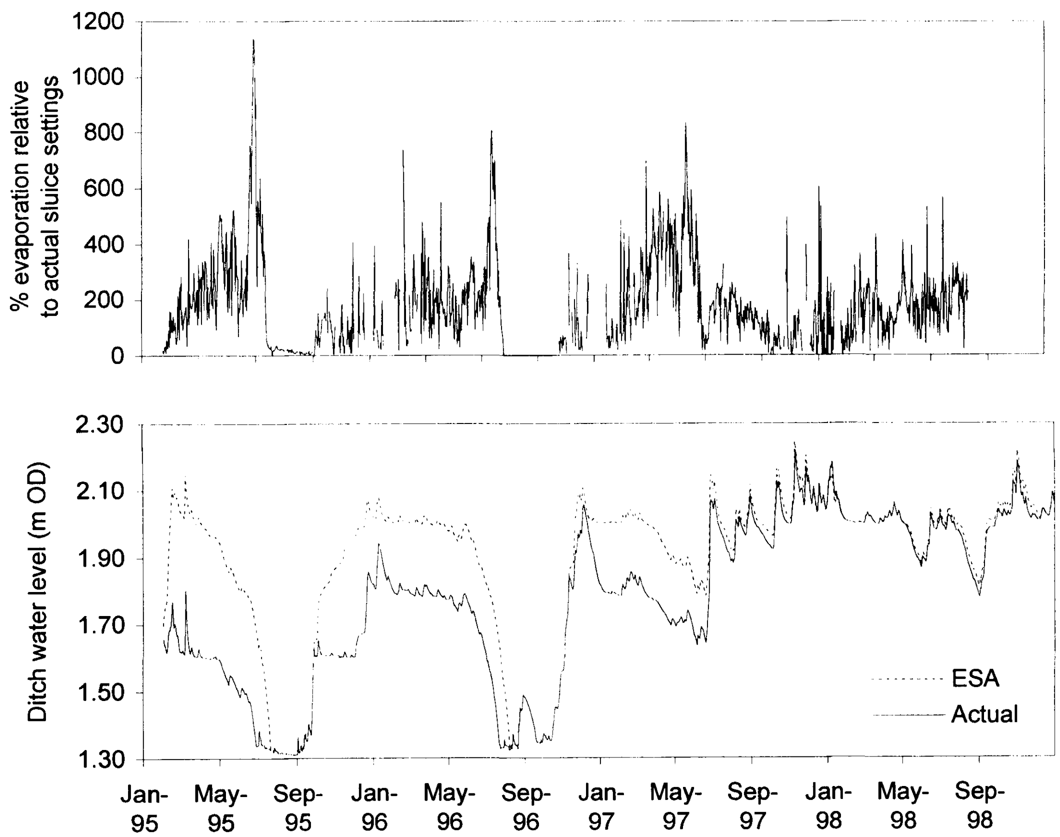


Figure 6.14. Volumetric losses by evaporation from the Field 2 catchment under ESA water level prescriptions relative to actual sluice settings.

Further detail regarding the effects of raising ditch water levels on evaporative loss is provided in Figure 6.14. In that figure, daily evaporative loss associated with the implementation of ESA water level prescriptions (used as an example of a scheme associated with higher water levels) are shown relative to evaporative loss on the SWT Reserve estimated from actual sluice settings 1995-1998. Whilst Table 6.16 shows that, on an annual basis, evaporative loss increases with increasing water levels, the analysis presented in Figure 6.14 serves to identify the times of year when raising ditch water levels has the largest influence on the ditch water balance.

Figure 6.14 shows that the largest increases in evaporative loss due to ESA prescriptions are predicted for 1995 and 1996. This is mainly because water levels prior to July 1997 were maintained at levels substantially below those prescribed by the ESA and CS schemes (Section 6.4.3). More mportantly, results presented in Figure 6.14 help to identify the spring months as a crucial period in the context of revised water level management strategies.. For all years considered, implementing ESA rescriptions leads to higher rates of evaporative loss than those apparent under actual sluice settings. The result is steeper summer recession curves (Figure 6.14.b) that lead to ditches drying out during dry summers, regardless of how high water levels have been maintained in winter. This observation helps to explain the limited overall impact of maintaining higher winter water levels on the frequency of water levels less than 1.40m OD in dry summers such as 1995 and 1996 (Section 6.4.4.1; Figure 6.12).

6.4.4.3. Providing suitable conditions for birds based on CS and ESA prescriptions

Model results have important implications for wetland management strategies that target wet grassland bird species on the Pevensey Levels. Model predictions indicate that substantially raising sluice levels has a negligible impact on overall water levels. As a result, predicted exceedances beyond the water level requirements of birds associated with the implementation of the CS or ESA schemes do not show large differences relative to each other. Exceedances beyond the water level requirements of birds associated with the implementation of the CS and ESA schemes on the SWT Reserve are shown in Figure 6.15. The largest changes relative to actual settings are in the magnitude of the exceedences during 1995 and 1996, although overall, the annual duration of exceedences are largely unaffected. Almost continuous negative exceedences are a feature of closing sluices or the implementation of actual, ESA and CS prescriptions on the SWT Reserve (Figure 6.15).

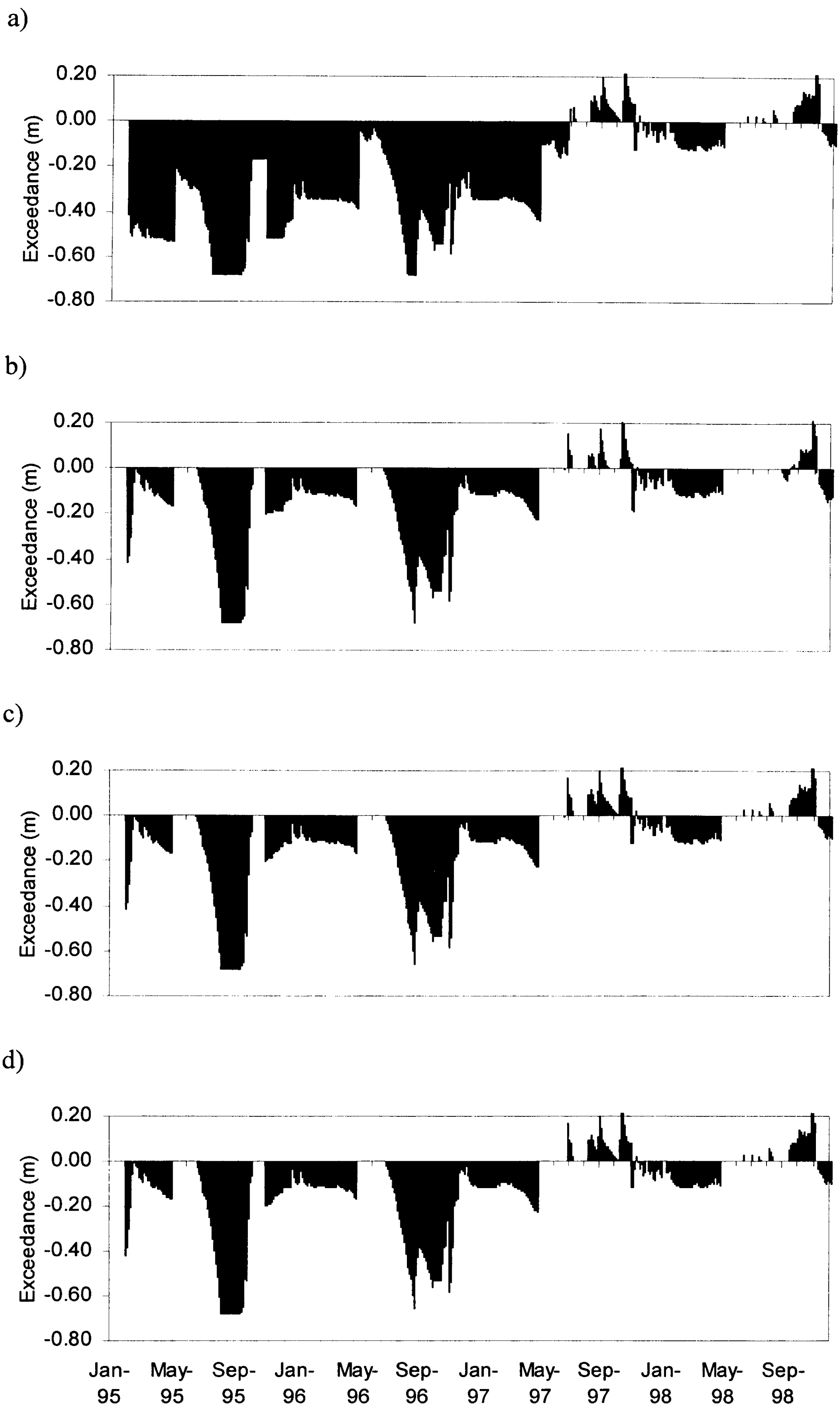


Figure 6.15. Exceedences beyond the requirements of birds due to the implementation of (a) Actual, (b) ESA Tier 3, (c) Countryside Stewardship water level management prescriptions, and (d) closing sluices on the SWT Reserve.

Model predictions indicate that raising winter sluice levels does not limit the large negative exceedences recorded during the summers of 1995 and 1996. As previously stated, these periods coincide with the drying out of the ditches on the reserve (Section 6.4.4.2). The onset of these conditions is not substantially altered by retaining higher winter water levels, due to associated increases in the evaporative rate (Section 6.4.4.2). The greatest changes correspond to the winter months, the period when the importance of inundation to wet grassland bird species is greatest (RSPB *et al.*, 1997). Negative exceedences recorded during these months were considerably smaller for the CS, ESA schemes and closing sluices altogether than for actual conditions. Model results indicate that for the entire four year period, implementation of ESA prescriptions will result in mean daily exceedences of -0.11m day^{-1} in winter. In wetter winters, such as that of 1997/98 however, mean daily exceedences are smaller (-0.06m day^{-1}). The highest mean daily winter exceedence recorded corresponds to 1995/96 when on average, water levels were 0.15m below the level required by birds.

Although the magnitude of all these exceedences are small, they are compounded by the fact that topographical surveys of the SWT Reserve described in Section 5.2.5 indicate that small changes to water levels have a large influence on the extent of inundation (Figure 6.1). For example, the mean daily exceedence recorded during the winter of 1995/96 is equivalent to a water level of 2.10m OD. At this water level, only 17% of the total Field 2 catchment area will be inundated (see Figure 6.1 and Table 5.5). These results clearly indicate the difficulties of satisfying the water level requirements of birds in drier than average years, although in wetter years the provision of large inundated areas remains a realistic objective. The mean daily exceedence recorded during the winter of 1997/98, although only 0.05m higher, results in the doubling of the inundated area. At a water level of 2.17m OD, 33% of the total Field 2 catchment area will be inundated, in close correspondence with the inundated area required by wet grassland bird species during the winter months (Section 6.3.3). Model predictions also highlight the need for flexible sluice management in areas where CS or ESA prescriptions are instated. On a number of occasions during the summers of 1997 and 1998, model results indicate that maintaining sluice levels in accordance to CS and ESA prescriptions results in positive exceedences (Figure 6.15), indicative of water levels in excess of those required by bird species, with potentially negative effects on nesting success and food supply.

6.4.5. CLIMATE CHANGE

The ability of wetland managers to satisfy the water level requirements of stakeholders on the Pevensey Levels is considerably influenced by climatic variability. This is very much the case with regards to current management, where sluice keepers must operate flexibly to ensure the supply of water to retain wet fences in the summer, and to provide sufficient flood storage capacity for the evacuation of excess winter rainfall (Section 2.4.6). Given the considerable media coverage regarding global climate change in recent years, members of the Pevensey Levels Study Group have frequently shown interest in the potential effects of climatic change on wetland hydrology and management. In particular, the drought years of 1995, 1996 and 2003 have raised awareness of water resource issues on the wetland, including the sustainability of existing, and proposed, water level management strategies. It is expected that climate change will have especially large impacts on water level management practices adopted during the summer months. This issue has been highlighted by the current difficulties of satisfying stakeholder requirements during dry summers (see Section 6.4.4) where even retaining higher winter water levels has a limited effect on the drying of ditches during drought years.

Although estimates vary, data provided by Global Circulation Models suggest that in Southern England, climatic change will be associated with net increases in temperatures of between 1.3 and 3.3^{o}C by 2050, depending on the emissions scenario adopted (Hulme and Jenkins, 1998). During the equivalent period, winter rainfall is predicted to increase by between 6 and 9% and summer rainfall to decline by between 3 and 19% (Hulme and Jenkins, 1998). Fewer data are available describing the impacts of climate change on evaporation. UKCIP (United Kingdom Climate Impacts Programme) data have quantified the impacts of climate change on the range of variables commonly employed to compute evaporation by the Penman formula (see Table 4.2). Budhyko (1980, cited in Nemec and Schaake, 1982) provides an estimate of the effects of climate change on evaporation, equivalent to a 4% per ^{o}C. According to Arnell and Reynard (1993), with credible assumptions about increases in radiation and reductions in humidity, the annual increase in evaporation will range from around 9% to 30% by 2050. Similar increases in summer evaporation of 40% by 2050 are proposed by the Department of the Environment for Southern England (DoE) (now Department of the Environment, Transport and the Regions, DETR) (1996). All of these data must however be treated with caution as they are at least a decade old.

Further limitations can be ascribed to the fact that numerous studies have actually shown that CO_2 enrichment can actually lead to a decrease in evapotranspiration, since biotic responses can differ greatly from those that consider temperature changes alone (Kuchment and Startseva, 1991, Watson *et al.*, 1996). Few allowances have been made for changes to plant stomatal conductivity resulting from higher CO_2 concentrations (DoE, 1996). Previous work has illustrated that such water efficiency gains could offset a substantial proportion of any climatically-induced increase in potential evapotranspiration (Arnell and Reynard, 1993). There is experimental evidence that some groups of plants use water more efficiently when CO_2 concentration is higher. Kimball *et al.* (1993) for example, found that spring wheat grown at 550ppm CO_2 had an evaporation rate 11% lower than wheat grown at 370 ppm CO_2. The response is also dependant on land use. Modelling studies by Kuchment and Startseva (1991), evaluating the impacts of climate change on evaporation from arable and grazed agricultural land in Russia using a soil moisture model, predicted changes in evaporation ranging from an increase of 41% to a 25% decrease, depending on land use and the scenario adopted.

Due to the contrasting evidence regarding the influence of climate change on evaporation, two approaches were employed to evaluate the impacts of climate change on the Pevensey Levels. In the first instance, the impacts of climate change on evaporation applied the data proposed by Budhyko (1980) to temperature change estimates predicted in UK-based literature. These predictions, when coupled to data describing percentage changes in rainfall as predicted by GCMs for equivalent years, were implemented within the PINHEAD model by perturbing input rainfall and evaporation time-series and re-running the model for different scenarios and years (*e.g.* 2020, 2050, 2080). In this case, seasonal predictions of both rainfall and evaporation were adopted by taking the months between June and August as the summer months, and the months between December and February as winter, as employed by UKCIP scenarios. Fixed percentage changes to rainfall and evaporation (as predicted by different scenarios) were then applied to these months. Data for intervening months were interpolated from summer and winter estimates of the percentage change to evaporation and rainfall in much the same way as the stage-discharge relationship for the estimation of sluice discharge in spring and summer transition derived for application within PINHEAD (see Section 5.3.4.3). A second method applied sensitivity analyses to evaluate the response of the field-scale hydrological system to changes in evaporation and rainfall.

Figures 6.16 and 6.17 show estimated changes to UK temperature and rainfall respectively for a variety of years and climate change scenarios. Implementation of these data within PINHEAD applied only estimates presented for the scenario labelled 'medium-low'. This choice accounted for climate change of a medium likelihood, rather than adopting the worst-case scenario. The impacts of a greater range of rainfall and evaporation extremes is evaluated in the context of model sensitivity. The specific temperature and rainfall changes implemented within PINHEAD are given in Table 6.17. As previously stated, predicted changes in temperature were employed to adjust evaporation time-series according to the data presented by Budhyko (1980). A 'no change' scenario, equivalent to actual water levels 1995-1998, was used as the control against which comparisons of the impacts of climate change could be made.

The impacts of climate change on water levels on the SWT Reserve are presented in a number of ways. In Figure 6.18.a, impacts are illustrated by the resultant water level time series. Figure 6.18.b shows the difference between actual water levels and those predicted by the model based on the climate change scenarios for 2020, 2050 and 2080. This enables the identification of times of year when impacts are greatest. Table 6.17 quantifies the effects of medium-low climate change predictions for the years 2020, 2050 and 2080 on the frequency of water levels in excess of 2.00m OD and less than 1.40m OD. This analysis has been undertaken using sluice settings for birds, equivalent to ESA Tier 3 water level prescriptions, and grazing, as well as actual sluice settings to evaluate the potential effects of climate change on other future water level management options that may be implemented on the Pevensey Levels wetland.

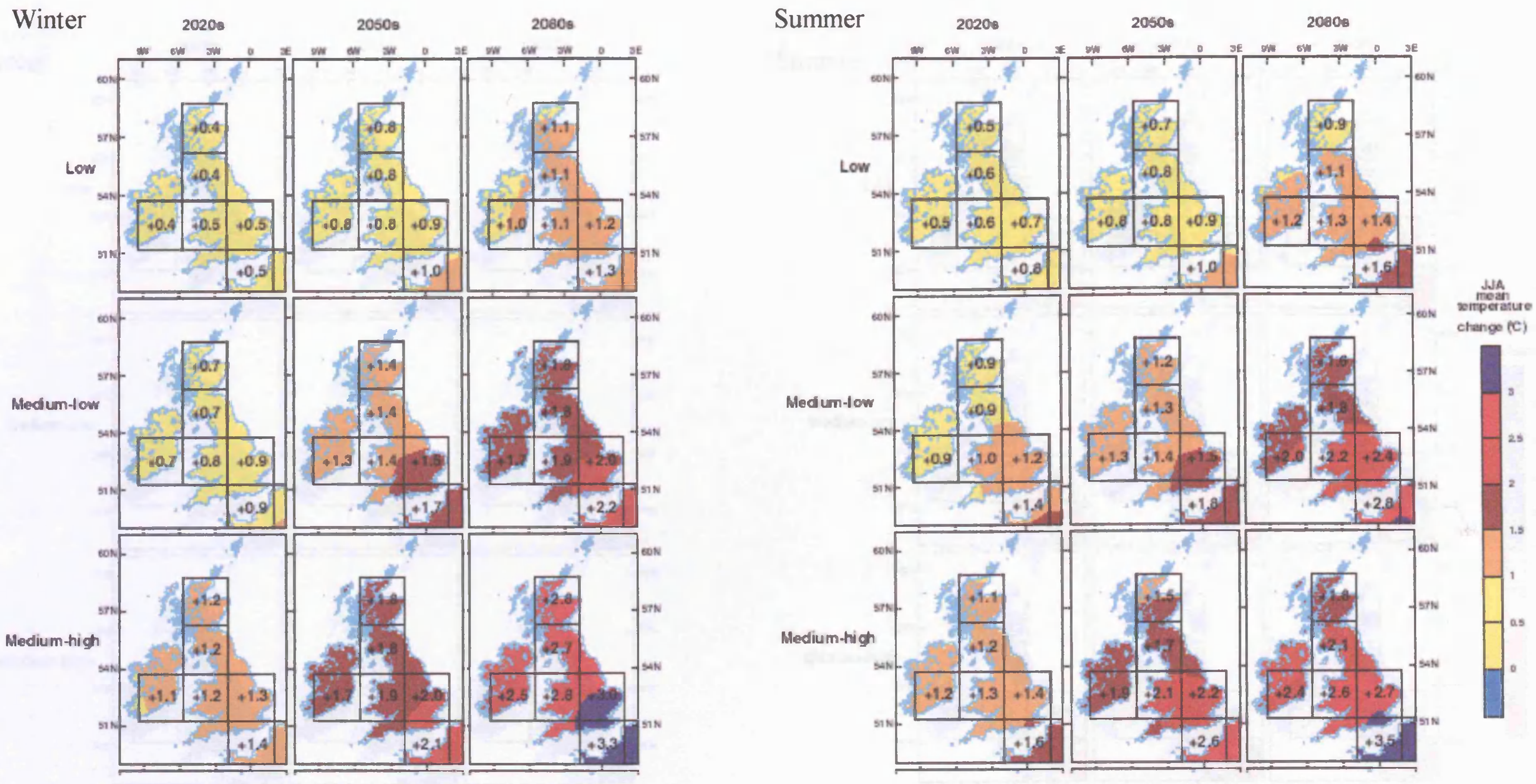


Figure 6.16. Changes to summer and winter temperatures for 2020, 2050 and 2080 under different climate change scenarios (Hulme and Jenkins, 1998).

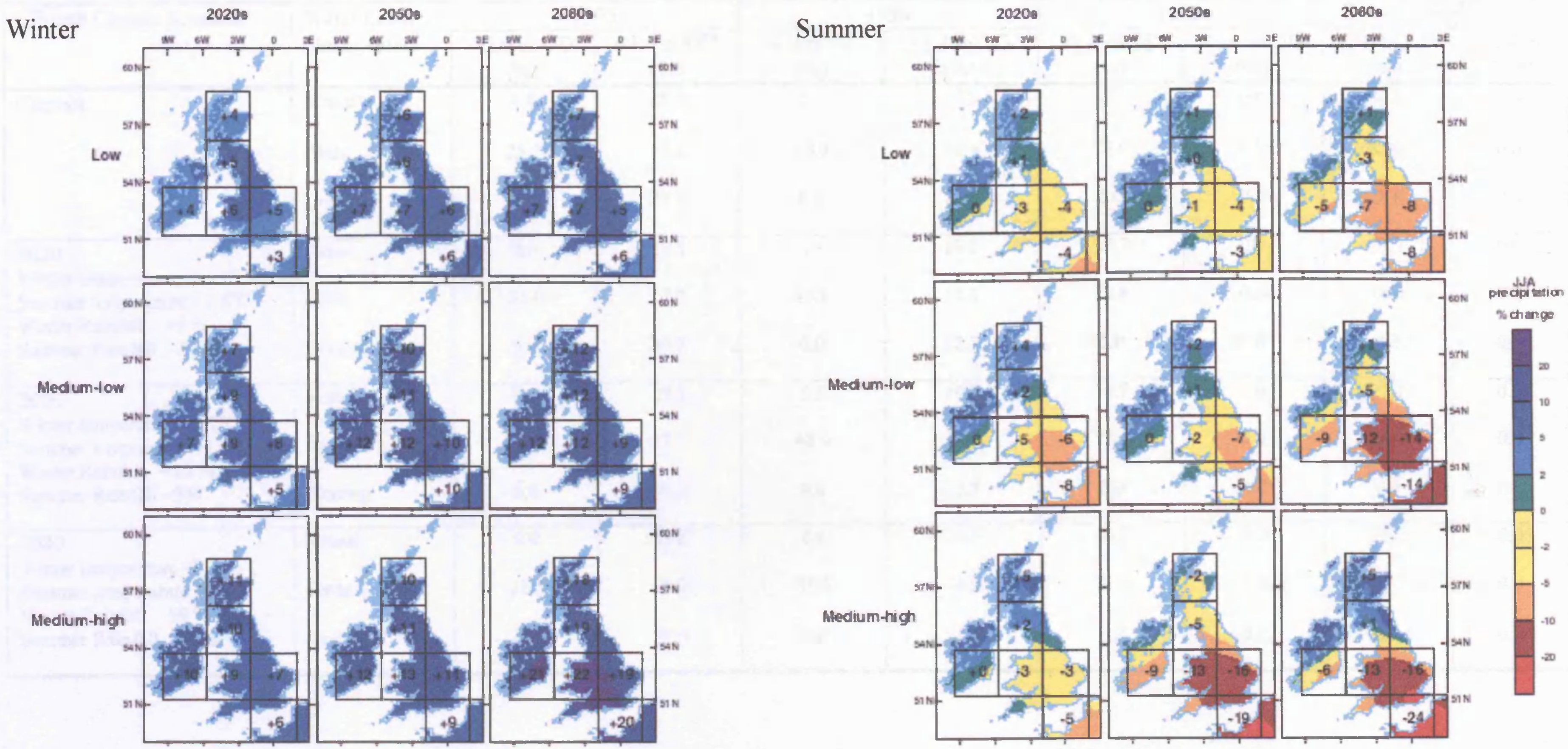


Figure 6.17. Changes to summer and winter precipitation for 2020, 2050 and 2080 under different climate change scenarios (Hulme and Jenkins, 1998).

Climate Change Scenario	Water Level Management	1995		1996		1997		1998	
		>2.0 m OD (%)	<1.4 m OD (%)	>2.0 m OD (%)	<1.4 m OD (%)	>2.0 m OD (%)	<1.4 m OD (%)	>2.0 m OD (%)	<1.4 m OD (%)
Current	Actual	0.0	26.9	2.7	17.5	38.1	0.0	80.8	0.0
	Birds	25.7	17.1	48.9	10.4	72.6	0.0	81.0	0.0
	Grazing	0.0	29.0	0.0	20.8	0.0	0.0	0.3	0.0
2020 Winter temperature +0.9°C Summer temperature +1.4°C Winter Rainfall +5 % Summer Rainfall –8%	Actual	0.0	27.5	4.1	19.1	37.3	0.0	77.7	0.0
	Birds	24.6	17.7	48.6	12.3	71.8	0.0	78.0	0.0
	Grazing	0.0	30.2	0.0	22.7	0.0	0.0	0.3	0.0
2050 Winter temperature +1.7°C Summer temperature +1.8°C Winter Rainfall +10 % Summer Rainfall –9%	Actual	0.0	27.5	5.5	19.1	36.7	0.0	77.7	0.0
	Birds	26.0	17.7	48.9	12.3	72.1	0.0	78.0	0.0
	Grazing	0.0	30.2	0.0	22.7	0.0	0.0	0.3	0.0
2080 Winter temperature +2.2°C Summer temperature +2.8°C Winter Rainfall +9 % Summer Rainfall –14%	Actual	0.0	27.8	4.6	20.5	36.7	0.0	77.2	0.0
	Birds	22.5	18.0	48.6	14.2	71.0	0.0	77.7	0.0
	Grazing	0.0	30.5	0.0	24.6	0.0	0.0	0.3	0.0

Table 6.17. Impacts of predicted climate change in 2020, 2050 and 2080 on the frequency of water levels greater than 2.00m OD and less than 1.40m OD under three different water level management strategies.

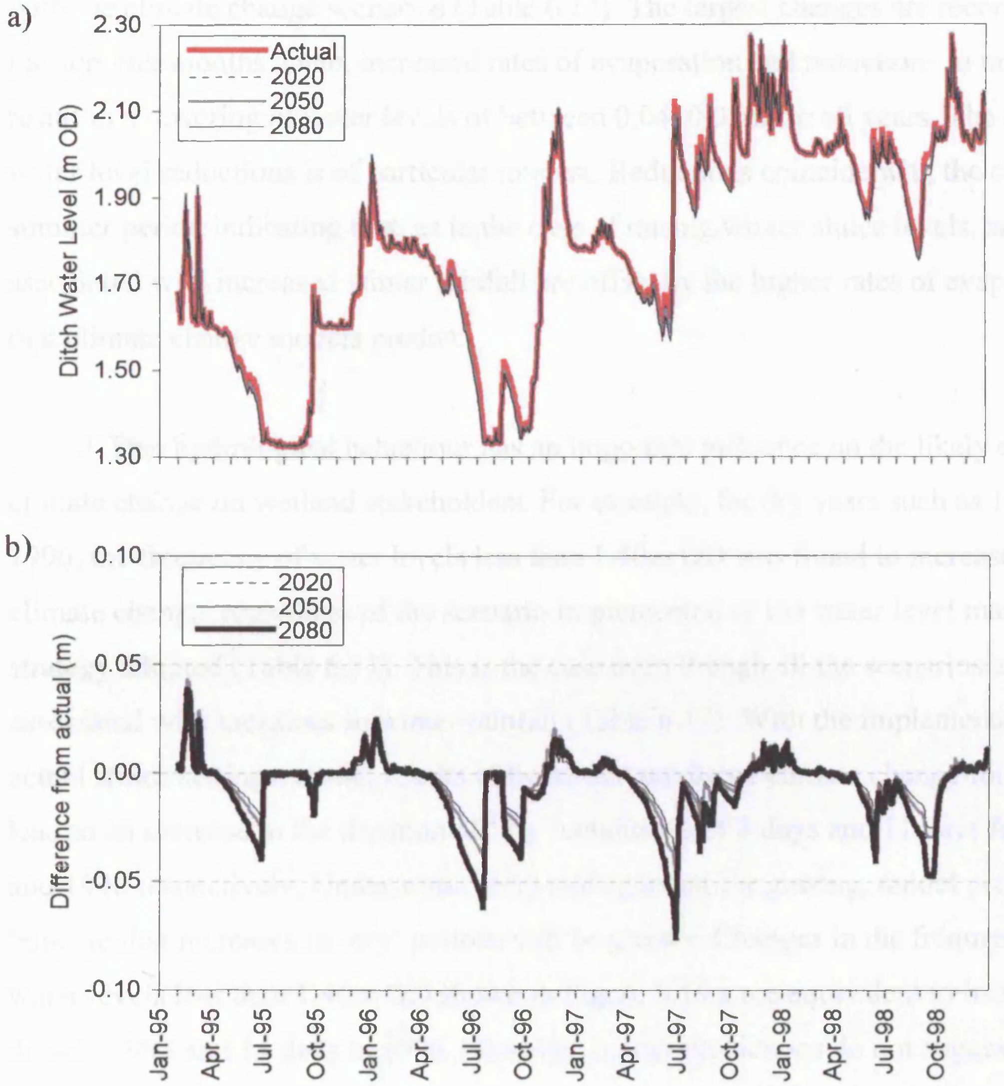


Figure 6.18. (a) Water levels predicted on the SWT Reserve from actual sluice settings and medium-low climate change predictions for 2020, 2050 and 2080 applied to the period between January 1995 and December 1998. (b) shows changes in water level due to climate change relative to actual water levels.

Model predictions indicate that climate change will cause small, subtle changes to water levels on the SWT Reserve (Figure 6.18.a). Changes are most clearly demonstrated as a function of water level change relative to water levels predicted based on actual climatic conditions (Figure 6.18.b). Small increases in water level are apparent for all winters during the study period due to the increased winter rainfall associated with the climate change scenarios (Table 6.17). The largest changes are recorded during the summer months when, increased rates of evaporation and reductions in rainfall result in a lowering of water levels of between 0.04–0.07m for all years. The timing of water level reductions is of particular interest. Reductions coincide with the crucial mid-summer period indicating that, as in the case of raising winter sluice levels, any gains associated with increased winter rainfall are offset by the higher rates of evaporation that climate change models predict.

This hydrological behaviour has an important influence on the likely effects of climate change on wetland stakeholders. For example, for dry years such as 1995 and 1996, the frequency of water levels less than 1.40m OD was found to increase with climate change, regardless of the scenario implemented or the water level management strategy adopted (Table 6.17). This is the case even though all the scenarios adopted are associated with increases in winter rainfall (Table 6.17). With the implementation of actual sluice settings, model results indicate that predicted climate change for 2080 will lead to an increase in the duration of 'dry' conditions of 3 days and 11 days for 1995 and 1996 respectively. Under water level management for grazing, model predictions indicate that increases in 'dry' periods will be greater. Changes in the frequency of water levels less than 1.40m OD shown in Figure 6.19.a are equivalent to increases of 5 days in 1995 and 14 days in 1996. However, model predictions do not suggest an increase in the frequency of inundation due to the higher winter rainfalls in areas managed for grazing (Table 6.17), probably because any increased water volume is evacuated through sluices. Similar impacts on the ability of wetland managers to satisfy the requirements of wet grassland birds are predicted by the model. Under ESA Tier 3 prescriptions, Table 6.17 and Figure 6.19.b indicate that predicted climate change by 2080 would reduce the frequency of the highest water levels in all years during the study period. Changes in the frequency of water levels greater than 2.00m OD shown in Figure 6.19.b are equivalent to reductions of 12, 1, 6 and 12 days between 1995-1998 respectively relative to actual climatic conditions.

These results indicate that climate change will compound the previously discussed difficulties posed by raising ditch water levels to create conditions suitable for birds. Table 6.18 summarises the influences of climate change on annual exceedence duration and SEVs for birds. The results refer to water level management accordant with ESA Tier 3 water level prescriptions. On an annual basis, predicted changes to rainfall and temperature result in an increase in the duration of exceedences (Table 6.18.a) with a concurrent effect on the associated SEVs (Table 6.18.b). Similar results are evident when the water level requirements of graziers are considered. Previous sections have highlighted the difficulties local farmers have faced in providing adequate grassland irrigation and maintaining wet fences during the summers of 1995 and 1996. Results presented in Figure 6.19.b indicate that an overall effect of climate change is to increase the frequency of such events. Given that characteristic ditch flora and the fen raft spider are dependant on a water level management regime for grazing it is presumed that such species will also be affected.

a)

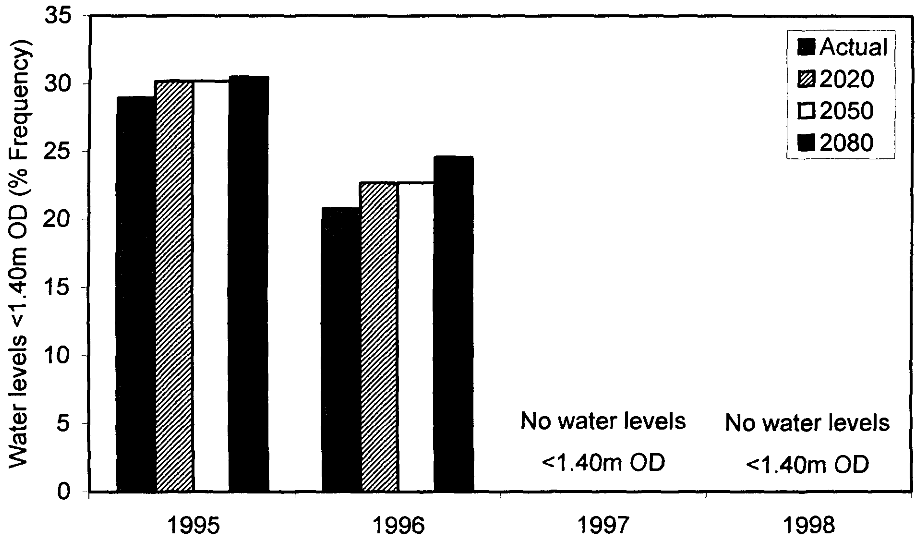
	1995	1996	1997	1998
Actual	-50.5	-58.0	-9.7	-8.3
2020	-52.9	-60.1	-10.6	-9.0
2050	-52.18	-60.1	-10.8	-8.9
2080	-53.63	-61.9	-11.9	-9.6

b)

	1995	1996	1997	1998
Actual	275	305	250	233
2020	275	308	258	232
2050	279	309	256	234
2080	273	312	265	229

Table 6.18. Predicted (a) Sum Exceedence Values (in m days) and (b) exceedence duration (in days) beyond the requirements of birds under due to ESA Tier 3 prescriptions and medium-low climate change scenarios for 2020, 2050 and 2080.

a)



b)

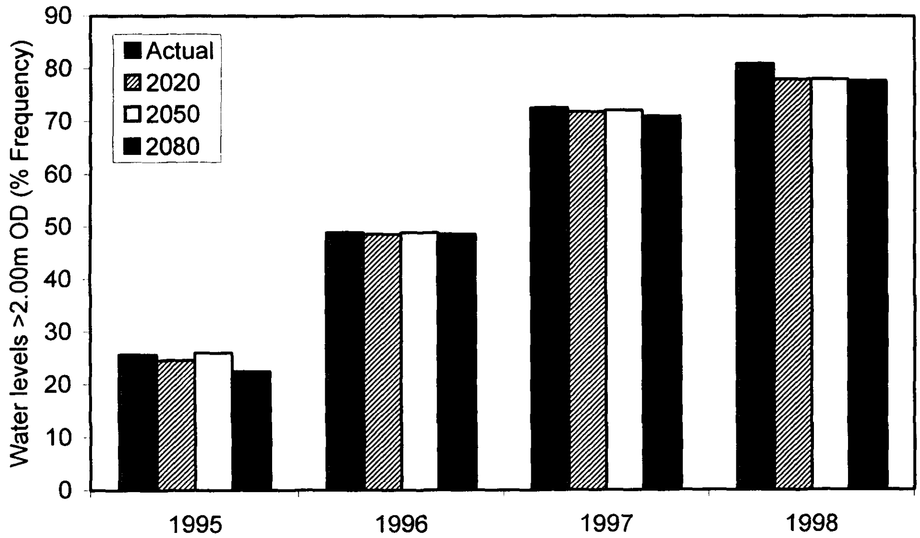


Figure 6.19. Graphical summary of Table 6.17. (a) Predicted changes to the frequency of water levels <1.40m OD due to climate change and a sluice management regime for grazing. (b) Predicted changes to the frequency of water levels >2.00m OD due to climate change and closing sluices.

6.4.6. INTEGRATING STAKEHOLDER WATER LEVEL REQUIREMENTS

The results obtained from the modelling approach applied in this chapter highlight the continuing difficulty of providing an integrated approach to water level management on the Pevensey Levels that may satisfy the multi-sectoral requirements of local stakeholders. The results are particularly applicable in the context of Water Level Management Plans (WLMPs). Analysis suggests that regardless of the water level prescriptions implemented by these plans, some negative impacts on local stakeholders will occur. For example it remains impossible to unify the water level requirements of birds and arable agriculture, although the requirements of key ditch flora and fauna can be routinely satisfied by implementation of 'traditional' water level management prescriptions for grazing (Figure 1.11). In this sense, the model has demonstrated that the current ditch water level regime is in tune with the water level requirements of the key flora and fauna which provide the wetland with its biodiversity value.

Section 6.4.3 has illustrated the value of WES in satisfying the water level requirements of a number of key stakeholders. The limited exceedences recorded beyond the requirements of graziers provide some explanation for the large percentage uptake of the scheme (Section 2.8.1). Results also indicate that WES water level prescriptions are close to the ideal requirements of key floral and faunal species on the wetland (see Figure 6.10). By limiting inundation however, they are not suitable for typical wet grassland bird species. However, the increased rates of evaporation associated with higher ditch water levels (Section 4.7.3 and 6.4.4.2) complicate the provision of water level conditions suitable for wet grassland birds, difficulties which are likely to be compounded due to climate change (Section 6.4.5). The continued uptake of the WES will rely on addressing some of the problems considered in Section 6.4.3 (*e.g.*. flooding of gateways), most of which can be resolved by unertaking field surveys prior to the implementation of a given water level management strategy. Flexibility will be an important component of any water level management strategy. For example, from a farming perspective, flexibility will be required to control increases in the frequency of inundation associated with the implementation of WES (see Section 6.4.3.1), events that can be averted by rapid management responses to storm events by either landowners or the operating authority.

These results clearly support the need for an alternative approach to the 'install it and forget routine' cited by Skaggs (1992). This is especially the case due to inter-annual climatic variability. Model results suggest that water level prescription,s and the sluice management regimes associated with them, should respond to the prevailing climatic conditions as well as the target stakeholder requirements. Although this is already a feature of the management of the Pevensey Levels wetland, it has on numerous occasions been stated that current manpower is insufficient to provide this required flexibility. Results suggest that any further changes to the hydrological regime will simply increase the pressures on the time spent by staff responding to the water level requirements of local landowners. The predictions of the impacts of climate change show that the degree of flexibility required is likely to increase in the future.

The specific water level management prescriptions that will be associated with WLMPs for the Pevensey Levels will at least be partially determined by changes to the agricultural subsidies received by farmers in the future. Management strategies that seek to enhance wetland habitat value for nature conservation objectives have consistently large impacts on the ability of farmers to maximise productivity, and hence on their ability to remain economically viable. Numerous farmers have highlighted that the current system of subsidies remains insufficient to maintain their way of life. This is becoming increasingly the case due to the continued perceived crisis within the agricultural sector. On the Pevensey Levels, this has led to a wide diversification of farming practices. Farmers on the wetland are involved in organic production, intensive dairy and arable farming, or traditional grazing. Turfing is also becoming an increasingly common practice (Joe Norris, Farmer, Pers. Comm.). These types of practices take place in an area where nature conservation interests are also considerable. Another factor that will dictate the future of water level management on the Pevensey Levels wetland will be the amount of water available for any proposed schemes. The rates of evaporation measured on the Pevensey Levels wetland, coupled to predictions regarding the likely effects of raising ditch water levels on this process, identify the difficulty of providing the large inundated areas required by the majority of wetland species. An important component of any water level management strategy will be detailed topographical and hydrological survey prior to its implementation. This will ensure that landowners are not affected by management practices on adjacent land.

CHAPTER 7

CONCLUSIONS AND RECOMMENDATIONS

7.1 Introduction

This thesis has considered various aspects of the hydrological functioning of the Pevensey Levels wetland, East Sussex. In common with most wet grassland habitats, the Pevensey Levels are a 'cultural' landscape (Section 1.1). Current hydrological functioning is a result of the progressive reclamation of the site from the sea since the Middle Ages (Section 2.2.1). On the Pevensey Levels, reclamation was achieved by 'inning' parts of the wetland within embankments and stabilising the shingle ridge that currently forms the southern boundary of the site (Section 2.2.1). Reclamation has had a profound influence on the configuration of the wetland drainage system and hydrological functioning at the catchment scale. This is particularly the case with respect to the Wallers Haven. This watercourse is the main surface water inflow to the site, draining the upland area to the north of the wetland. During the 17th Century, the course of the Wallers Haven was altered (Figure 2.2, Section 2.2.1) to allow the more effective evacuation of flood waters from the wetland. In doing so, the wetland catchment was effectively split into two distinct surface water systems, the Wallers Haven to the east and the Pevensey Haven to the west (Section 2.4.1). Subsequent drainage efforts throughout the 20th Century included the installation of sluices and pumping stations and the construction of ditches and grips, field scale drainage features used to evacuate water from the field centre into the ditches (Section 2.2.2).

A direct result of this drainage history has been the compartmentalisation of wetland hydrological functioning. Three distinct spatial scales of hydrological functioning are apparent on the Pevensey Levels: the field scale, the pumped sub-catchment scale and the wetland scale. However, the arrangement of the drainage system is such that these three scales are linked. The design of the drainage system is intended mainly to maximise agricultural production, the main land use on the site since reclamation. Between upper and lower limits, the landowner can control water table levels within the wetland by managing the penning-board type sluices that are commonly present where ditches inter-connect. There are over 250 structures for water level management on the wetland (Section 2.4.6), which allow landowners to either connect to, or isolate themselves from, a pumping station. There are eight operational pumping stations across the wetland (Figure 2.7) and all discharge into high level embanked channels (Section 2.4.3). Water levels in the embanked channels are controlled by large gates located at the downstream end of each watercourse. These gates are managed on a seasonal basis, essentially to retain water in the summer (to allow 'feeding' of the lowland area for crop irrigation, drinking water for depastured stock and 'wet fencing') and evacuate runoff in winter (Section 2.4.5), thus reducing the risk of flooding and waterlogging. During winter, a particular problem is that the wetland remains tide-locked for a large proportion of each tidal cycle. Most of the wetland is below the mean high tide level. Consequently, embanked channels have been engineered to provide flood storage for winter runoff.

This hydrological description provides the background for the work that has been undertaken as part of this thesis. To understand the hydrology of the wetland has necessarily required hydrological studies at the three spatially-distinct scales of functioning identified on the wetland. A key focus of all the studies undertaken has been the consideration of wetland hydrology relative to the requirements of wetland stakeholders. On the Pevensey Levels, hydrology is frequently emphasised as the source of conflict between different stakeholders. This is because the water level requirements of different stakeholder groups (nature conservation, agriculture and flood defence), are different. For example, wetland biota require extensive areas of surface inundation, but such practices can have a detrimental effect on agricultural productivity (Section 2.4.6).

The issues considered in this thesis have been largely identified by discussions with members of the Pevensey Levels Study group, a coalition of stakeholder representatives that meets on a bi-annual basis (Section 2.8.3). By addressing issues relating to the management of the wetland within a committee-type framework, the Pevensey Levels Study Group can be considered an excellent example of how sustainable management of a wetland can be effectively achieved. Due to the overriding influence of hydrology on all wetland stakeholders, and the different hydrological requirements of each of the groups, the decision-making process has frequently required scientific opinion to address the feasibility of proposals and the potential impacts of proposals on other members of the group. Work undertaken in this thesis is therefore essentially a response to information requested by different members of the group throughout the duration of the project.

7.2 Water level management on the Pevensey Levels

A key feature of the historical, current and future management of the Pevensey Levels wetland is the management of water levels to satisfy all stakeholder interest groups. For farmers, a distinct seasonal approach is required to maintain an economically viable business. For grazing, which has historically been the main agricultural practice on the Pevensey Levels, this requires water levels some 0.5m below field level between April and October and 0.7 m below ground during the autumn and winter months (Figure 1.11, Section 1.6.3). This hydrological regime ensures that during summer the grass crop remains irrigated, sufficient drinking water is available for de-pastured stock, and field boundaries are maintained by a network of wet fences. Lower water levels maintained in the autumn and winter reduce the risk of waterlogging and surface inundation of fields, which can have a large influence on the productivity of the grass crop.

The implementation of this hydrological regime has been largely possible due to the installation of a pump-drainage system. Pump-drainage has allowed re-seeding of wetland grasses with more productive species and has also enabled the expansion of arable farming in the area (Section 2.2.2., Figure 2.3). However, the hydrological regime required for agriculture is essentially an inversion of the 'natural' hydrological regime, where field inundation during winter and early spring is followed by a progressive reduction in ditch water levels throughout late spring and summer in response to increased evaporation. Alteration of this 'natural' hydrological regime has been perceived as causing the ecological degradation of the wetland. Numerous studies support this assertion, including studies on bird numbers, and the distribution of rare flora and fauna across the site (Section 2.7).

In response to this ecological degradation, management of the site throughout the last decade has been targeted towards the progressive restoration of 'natural' hydrological conditions on the site. A key focus of the management approaches which have been implemented has been a strong commitment to revising water level management strategies across the wetland. Since 1991, English Nature have been involved in the promotion and implementation of a voluntary wetland management scheme. The Wildlife Enhancement Scheme (WES) pays landowners to adopt environmentally-sensitive farming practices, including controls on the timing and intensity of grazing and the application of fertilisers and pesticides (Section 2.8.1). A key management prescription relates to water level management in wetland ditches. Between January and August, ditch water levels should be maintained at no less than 0.3m below field surface, and at no more than 0.6m below field surface at other times of year (Table 2.13).

In conceptual terms at least, the WES is a success. The scheme promotes a water level management strategy that is truly integrative in nature. It attempts to address the balance between nature conservation and agriculture on the site by not advocating surface inundation, which is the main concern for farmers, but encourages high water levels in ditches as required by nature conservation. For this reason, a large number of local landowners have been signatories to the scheme, especially since implementation has coincided with changes to the profitability of farming, including the Bovine Spongiform Encephalopathy and Foot and Mouth (BSE) crises, and changes to the prices of agricultural commodities due to reforms of the Common Agricultural Policy.

However, the scheme has not been without its problems (Table 2.14), many of which are applicable not just to the Pevensey Levels and the WES, but also to wider issues associated with wetland management in the UK. For example, the WES has been criticised by nature conservationists for not openly promoting surface inundation and by farmers because it does not incorporate grants for capital funding of changes to the drainage and farm infrastructure (Section 2.8.1). Key impacts on farmers have been the inundation of gateways, and the submergence of field drains leading to the increased frequency and duration of water logging, with effects on farm productivity and profitability. When the WES was drawn up no account was taken of the available water resource in the Levels system (Douglas, 1993). Water balance assessments previously conducted, reviewed in Section 2.5, have suggested that the area under the influence of higher ditch water levels should be limited due to the scarcity of water resources. For local water resource managers this is an issue of particular importance. The Pevensey Levels is located in one of the driest parts of the UK (Section 2.4.2) and represents an important source of public water supply for the expanding towns of Hailsham, Polegate and Eastbourne (Section 2.4.7).

7.3 Review of methods

To address the issues of importance to stakeholders on the Pevensey Levels, four distinct approaches of scientific enquiry have been adopted in this thesis:

- The collation and analysis of available hydrological data for the site;
- The collection of data and development of methods to quantify the hydrological parameters for which data is lacking;
- Detailed analysis of the hydrological character of the site at the field, pumped sub-catchment and wetland-wide scales, and
- Development of operational models of the hydrology of the Pevensey Levels to address issues of interest to wetland stakeholders, to aid in the decision-making process.

A recurring concept is that of the wetland water balance. The wetland water balance approach has been applied at the three scales of hydrological functioning identified on the Pevensey Levels wetland to address key issues of management importance. This has been possible due to the availability of a broad variety of data describing the hydrology of the site, historically collected by wetland managers to inform agricultural and water resource management. In terms of data availability, the Pevensey Levels do not generally conform with the suggestion by Beran (1982) that data availability is routinely a control on the effective management of wetland habitats (Section 1.1). Rainfall, climatic data, surface water inflows, channel water levels (*e.g.* embanked channels), pump drainage system functioning (water levels and hours pumped) and surface water abstraction have all been routinely monitored by the Environment Agency and its predecessors. Data describing hydrological functioning at the field scale were also available. In February 1995 a network of dipwells, piezometers and water level recorders was installed to evaluate the influence of raising ditch water levels on field water table levels in the SWT Reserve, in the central, gravity-drained part of the wetland (Figure 3.32). However, data describing losses to sea, lowland feeding during summer, and field-scale water levels in pump-drained areas were not available. Methods used to estimate each of these parameters within wetland water balance calculations are described in Sections 3.4.1 (Losses to sea and feeding) and Section 3.5.3 (water levels in pumped drained areas).

Of the issues described by Beran (1982) discussed in Section 1.1, the most important constraint to this study was the variety of measurement intervals employed for different hydrological parameters monitored on the Pevensey Levels wetland. Rainfall, river flow and climatic data are all collected on a daily basis. Water level and pumping station records are recorded on a roughly weekly basis. However, water level and pump hour data for different pumping stations are not necessarily coincident: the day when each is visited is different. For example, abstraction data were only available on a monthly basis, and although field-scale ditch water level data were collected continuously, water table levels could only be measured on a fortnightly basis.

As a result, for catchment-scale water balance calculations, a monthly time-step was employed. Available hydrological data were used to develop a wetland water balance for the period between January 1995 and December 1998 that quantified all the components of the hydrology of the Pevensey Levels. The wetland water balance incorporated all pump-scale data to provide an assessment that is semi-distributed in nature. The water balance was implemented as a fully operational spreadsheet model, capable of addressing the key concerns raised by local stakeholders, most notably:

- The evaluation of the importance of abstraction at the catchment-scale;
- Quantification of losses to sea through tidal sluices;
- The importance of the balance between rainfall and evaporation across the wetland;
- Water resource availability between years and at different times of year, in order to address the sustainability of abstraction and water level management strategies.

The use of the period between 1995 and 1998 was especially advantageous as it included years that were drier than average (1995 and 1996) and others that were wetter (1997 and 1998), allowing an assessment of management issues under a range of climatic conditions.

Field-scale studies also considered the period between 1995 and 1998, though on a more intensive temporal scale of assessment. Hydrological studies on a daily basis were possible because ditch water level, rainfall and evaporation data were available at this time interval. Because the area where field scale hydrology was monitored was not connected to the pump drainage system, it was not influenced by the monitoring interval implemented at the wetland pumping stations. Ditch water level and water table monitoring was complemented by the installation an Automatic Weather Station and a Hydra mkII between the summers of 1996 and the end of the monitoring period in late 1998 (Section 4.6.2). In the case of the Hydra, data were only collected during the summer period (early June to late September). This was considered the crucial period in terms of evapotranspiration, and also because the device does not function during periods of rainfall. Field scale hydrological processes for which data were not available included sluice discharge, rainfall-runoff relationships, and the interaction between surface water (ditches) and groundwater (in-field water tables). Methods for quantifying all of these processes were provided by analysis of ditch and shallow groundwater level data.

All field scale data were implemented within an operational spreadsheet model called PINHEAD (Physically Based, Integrated Hydro-Ecological model for the Assessment of Ditch systems; Chapter 5). PINHEAD uses the water balance approach to simulate ditch water levels using data commonly collected in wetland areas. Input data required by PINHEAD include:

- The location of sluices, blocked ends, gateways, roads and embanked channels to enable ditch catchment delineation;
- Data describing the dimensions of ditches in the target system (cross-sectional dimensions and the total ditch length in the catchment);
- Topographical data to evaluate inundation storage;
- Sluice level management data;
- Climatic input data (rainfall, evapotranspiration, soil moisture deficit);
- Ditch water level data for model calibration and verification.

The key objective of the development of PINHEAD was its ability to predict changes in water levels associated with various water level management options (mainly by changing sluice level management strategies) and to estimate the effects of these changes on stakeholders on the Pevensey Levels wetland. For this reason, this primarily hydrological model incorporates a hydroecological sub-model (Section 6.2) that compares ditch water level predictions with data describing the water level requirements of target stakeholders. In PINHEAD, these are then related to predicted ditch water levels to quantify the extent, duration and frequency of either exceedances beyond or below these requirements. Data describing the water level requirements of local stakeholders were obtained by a combination of methods, including literature reviews, but most importantly by discussions with local stakeholders who provided the most reliable and locally-applicable information for implementation within the model.

The model was then interrogated to address key issues of importance to local stakeholders at the field-scale, most notably:

- The effects of raising ditch water levels on evapotranspiration (Section 6.4.4.2);
- The volumes of water required to raise ditch water levels at different times of year to levels coincident with WES, Countryside Stewardship, Environmentally Sensitive Area Scheme or Water Level Management Plan prescriptions (Section 6.4.4);
- The effects of water level management for wetland biota on farmers (Section 6.4.3);
- The effects of agricultural water level management on rare wetland flora and fauna (Sections 6.4.1 and 6.4.2);
- The different approaches to management required during dry and wet years for farmers and nature conservation (Section 6.4.1 and Table 6.12);
- Evaluation of the potential impacts of climate change on the wetland (Section 6.4.5).

In many cases, field-scale studies were required to inform wetland-wide water balance calculations, illustrating the fore-mentioned inter-dependence of the various spatial scales of hydrological functioning on the wetland. Field-scale results implemented in a wetland-wide framework included:

- The use of field-scale evapotranspiration data within the catchment-scale water balance;
- The use of volumetric estimates of water required to raise ditch water levels to provide an assessment of the sustainability of raising ditch water levels wetland wide;
- Comparison of the volumes of water available in embanked channels at different times of year with increased water demand associated with raising ditch water levels.

7.4 Principal findings

In this section, the principal findings presented in this thesis are summarised. The issues considered were mainly identified during discussions of the Pevensey Levels Study Group (Section 2.8.3). Uncertainties associated with any of the conclusions drawn are identified. Later sections identify recommended future work to be undertaken on the Pevensey Levels, and on wet grasslands in the UK, to further confirm the findings presented in this thesis.

7.4.1 THE CATCHMENT WATER BALANCE

Catchment-scale water balance assessments have highlighted the key roles played by rainfall and evaporation in terms of overall water availability, a feature of the hydrology of wet grasslands previously noted by Hollis and Thompson (1996), Gilman (1989, 1990) and Cook and Moorby (1993). Rainfall across the site decreases in a roughly west to east direction (Section 3.3.1.2) and, at the catchment-scale, accounts for at least 40% of all wetland inflows in any month, although during winter this value frequently exceeds 60% (Figure 3.44.a). During winter, inflows from the Wallers Haven can approximate the contributions of rainfall, although on average they are 71% of monthly rainfall contributions (Section 3.7.1).

Inflows from sewage treatment works (STWs) are negligible on an annual basis (Table 3.16), although during dry summers they represent 20 % of all wetland inflows (Figure 3.44.a). The importance of STW discharges is greater when considered in a spatially distributed manner. Evidence has been provided by analysis of data collected at the Rickney pumping station. This pumping station drains the Horseye and Down pump-drainage system to which the Hailsham South STW discharges. Available data suggest that the functioning of the Rickney pumping station is considerably influenced by the STW discharges from Hailsham South STW (Section 3.5.4, Figure 3.30.f), with considerable water quality implications for this sub-catchment, and areas downstream of the pumping station. It is likely that areas connected to the Hurst Haven (to which the Hailsham North STW discharges) may also be subject to changes in water quality.

The largest outflows from the wetland on an annual basis are associated with evapotranspiration and evaporation (Section 3.7.1, Table 3.16). During the summer months (May-September), water losses by this process represented up to 80% of all outflows from the wetland (Figure 3.44b). Losses to sea from the wetland during winter accounted for equivalent proportions of water lost by evapotranspiration and evaporation in the summer (Figure 3.44b), although on an annual basis losses to sea were smaller in volumetric terms (Table 3.16). In contrast, when considered at the catchment scale, abstraction was negligible and rarely exceeded 15 % of all wetland outflows (Section 3.7.3 and Figure 3.44.b).

Field-scale studies have identified groundwater seepage (the movement of water from ground to surface) as a primarily winter process, with recharge dominant in the summer (Section 3.6.3). This is expressed as a negative hydraulic gradient in summer (ditch water levels are higher than water table levels) and a positive gradient in winter (water table levels are higher than ditch water levels)(Figure 3.37), a trend observed in other wet grassland wetlands in the UK (Section 1.6.4). However, calculations based on Darcy's Law suggest that such interactions are negligible in volumetric terms. Indeed, both seepage and recharge have been found to be at least two orders of magnitude less than the volumes represented by rainfall and evapotranspiration (Table 3.14). These results suggest that on the Pevensey Levels, and potentially in other clay-dominated wetland areas, seepage and recharge can be omitted from water balance assessments. This provides support for the treatment of the phreatic and surface water components of the local hydrological cycle as two separate entities.

7.4.2 THE RELATIONSHIP BETWEEN DITCH AND WATER TABLE LEVELS

The limited rates of exchange between ditches and fields highlight the difficulties of providing the water table conditions required by rare flora and fauna by raising ditch water levels alone. Calculations based on available estimates of hydraulic conductivity (K) for the site provided by Armstrong (1998) support suggestions by Douglas (1993) that water level management strategies such as WES will not deliver sufficiently different soil water regimes to those currently apparent wetland-wide. Based on the mean value of K on the SWT Reserve (0.057 md^{-1}), under steady-state conditions, it would take 1535 days for a water level set in a ditch to come into equilibrium with the water table in the centre of a field 175 metres wide (Section 3.6.2). This was the mean width of the fields monitored. This is supported by observations of ditch and water table levels. In most cases, the sphere of influence of the ditch was limited to dipwells located 2m and 5m from the ditch (Figure 3.35). At distances greater than 5m, water table variations were more closely related to the balance between rainfall and evaporation in the preceding period than to water levels in the ditch (Figure 3.35).

Results highlight that inundation of field surfaces is required to provide high water table levels. Raising ditch water levels to levels close to the field surface serves only to wet up ditch margins. Based on the *K* of local soils, shallow inundation water would take only 35 days to travel through 2.00m of clay, the typical thickness of the clay layer on the wetland (Section 2.3). If shallow surface flooding can be maintained for over one month each winter, fully saturated soils in spring are therefore a realistic objective, a condition is favoured by wet grassland species (RSPB *et al.*, 1997).

Evidence for more rapid water movement was available from peat water level data (Section 3.6.4). Peat commonly underlies the clay on the Pevensey Levels wetland. Although, seasonal trends in the relationship between water levels in the peat and ditches replicated those of clay dipwells (higher than the ditches in winter and lower in summer), in Field 3, water level variations in the peat at opposite ends of the same field were closely equivalent (Figure 3.40), indicating the equilibrium of water levels over extended distances. For Field 3, there was also a close relationship between peat and ditch water levels suggesting that in some parts of the wetland, the top of the peat layer and the bed of the ditch were coincident. However, the connectivity of ditches and the peat layer is likely to be spatially variable, as highlighted by the limited correspondence between peat water levels and ditch water levels in Field 2 (Figure 3.41).

7.4.3 WATER AVAILABILITY FOR REVISED WATER LEVEL MANAGEMENT

The ability for wetland managers to provide higher water levels than those currently implemented across the site is dependant on water resource availability at the catchment scale. On the Pevensey Levels, feeding of lowland channels by operation of sluices on embanked channels has traditionally been used to raise water levels in the lowland ditch network (Section 2.4.5). The precise magnitude of lowland feeding possible is however dependant on surface water storage in the embanked channels. Embanked channel storage is a combined function of the management of gates commonly located at their downstream ends, and the seasonal pattern of inflows and outflows. On the Pevensey Levels, there is a distinct seasonal trend in water resource availability. For all years during the study period, outflows exceed inflows between March and September (Figure 3.46.b). An exception was 1998, when the period of deficit began later, in June.

A comparison between the net water balance between 1995 and 1998 on a monthly basis, relative to the estimated demand associated with a variety of wetland restoration schemes, has been shown in Figure 3.46.a. The results illustrate the difficulty of supplying sufficient water to achieve water level prescriptions associated with both the ESA scheme and the WES in dry years such as 1995 and 1996. In wetter years however, such as 1997 and 1998, supplying sufficient water to achieve WES and ESA prescriptions remains a realistic target.

One potential option to provide extra water during dry years is to increase embanked channel storage to enable more extensive lowland feeding during times of water scarcity. The start of the net hydrological deficit in April coincides with the traditional timing of the reversion of wetland hydrological management to summer settings (Section 2.4.4). Results suggest that the reversion to summer conditions will have to take place earlier if additional water to supply wetland restoration strategies is to be retained (Section 3.7.2). Catchment-scale studies however indicate that reverting to summer conditions in March or even as early as February have a negligible influence on the balance between inflows and outflows during summer. For the years 1995 and 1997, reverting to summer settings on the Wallers Haven in February only reduced the period of net water resource deficit by one month, highlighting the need for storage during autumn/winter, an approach that would require an increase to the current storage capacity of the Wallers Haven. Results shown in Figure 3.50 illustrate that the storage of water required by the WES and ESA schemes would generally exceed current storage capacity, especially during the winter months, thus increasing the risk of flooding of bankside land beyond acceptable levels.

Equivalent results were obtained when storage in lowland, field-scale areas were considered. As in the case of embanked channels, the reversion to summer settings in field scale channels traditionally takes place in April (Section 2.4.6). Closing sluices earlier in the year had a limited impact on the frequency and duration of the period of water resource deficit at the field scale in dry years. Closing sluices in March reduced the frequency of water levels less than 1.40m OD (equivalent to 'dry' conditions) by only 4.1 % in 1995 and 6.5% in 1996 (Table 6.12). Closing sluices in February reduced the frequency of 'dry' conditions by 9.0 % in 1995 and 14.3% in 1996 (Table 6.12).

7.4.4 DITCH WATER LEVELS AND EVAPOTRANSPIRATION

Difficulties associated with limiting the onset of dry conditions in lowland ditch systems can be ascribed to evaporation and evapotranspiration, a process which at the catchment scale has been identified as the most important outflow from the Pevensey Levels on an annual basis (Section 3.7.1). A crucial finding associated with field-scale evaporation and evapotranspiration monitoring is that the rate of evaporative loss is partially determined by ditch water levels (Figure 4.7). During inundated conditions, rates of actual evapotranspiration (AET) can be up to 20 % greater than those estimated based on estimates of potential evapotranspiration (PET) alone. The validity of this relationship is supported by the fact that unity (AET=PET) occurs at bankfull conditions, when the wetland, in conceptual terms at least, most approximates Penman's idealized grass surface with a plentiful supply of water (see Section 4.2).

Results highlight the need for accurate estimates of evapotranspiration for wetland management, especially where management targets are associated with the provision of surface inundation, since this practice will increase evaporative loss. This effect should therefore be considered in water resource calculations undertaken for feasibility studies of raising ditch water levels in wet grasslands and wetlands more generally. For the calculation of evapotranspiration on the Pevensey Levels, and potentially in other wet grassland areas, AET can be inferred from tank evaporation data and ditch water levels, expressed as a function of the mean field level, as a surrogate of water availability. Equations to calculate AET from ditch water level data and other PET estimates have been provided in Table 4.11. For the Pevensey Levels, the relationship between AET, PET and water availability was not related in the manner suggested by Morton (1983) (see Figure 4.1.b), but akin to that proposed by the MAFF (1967), CROPWAT and MORECS models (see Figure 4.1.a). These latter models are traditionally employed in operational practice in the UK. However, all these models state that the rate of AET cannot exceed PET, which may lead to the under-estimation of evaporative loss in semi-inundated wetland areas. This will result in inaccuracies within water balance calculations in wetland areas.

An assessment of the influence of different evaporation and evapotranspiration estimates on water resource calculations has been undertaken in Section 4.8. Results shown in Figure 4.13 illustrate that, under in pump-drained parts of the wetland, use of a tank coefficient of 0.88 as proposed by Kadlec (1989) for dyked wetlands, over-estimates ET losses. On average, annual ET loss from the wetland between 1995 and 1998 was 60% of that calculated using this coefficient. A direct result of the adjustment of ET estimates based on ditch water levels was to reduce the monthly residual associated with wetland water balance calculations (Figure 4.13) by 92% (Section 4.8). This is an indication of the improved accuracy of this approach. Over-estimation by the traditional method is because, in the catchment-scale model, water levels wetland-wide are calculated from levels measured at the pumping stations (by assuming all ditches within each pumped sub-catchment are connected). Broadly speaking, the electrode levels implemented in pumping stations approximate the water level requirements of grazing. Evapotranspiration studies described in Chapter 4 suggest that at these water levels, a coefficient of less than 0.88 (the coefficient traditionally employed) is required for accurate estimation of catchment evaporative losses (Section 4.8).

Results indicate that the enhanced evaporative rates associated with inundated conditions and/or higher water levels will have a considerable influence on the ability of wetland managers to achieve water level targets. Field-scale modelling studies have shown that raising sluice levels in winter to achieve spring and early summer inundation has a limited effect on water level trends in late summer, particularly in dry years (Section 6.4.1). In dry years such as 1995 and 1996, raising sluices to levels akin with ESA prescriptions only serve to reduce the number of days during which ditches remain at their dry level by 9.5 and 9.9% respectively relative to actual conditions (Section 6.4.1). The over-riding influence of evapotranspiration on the field scale water balance in spring and summer is further emphasised by the limited effects of closing sluices altogether, a simulation that seeks to establish the effect of isolating the lowland ditch system from the pump-drainage system entirely. Relative to ESA prescriptions, this results in an increase of water levels greater than 2.00m OD (Figure 6.13.b) of 7.5% in 1995 and only 1.9% in 1995 and 1996. The duration of dry conditions is only reduced by one day in 1995 and 1996 (Figure 6.13.a), because any gains in water levels in late spring/early summer are offset by higher rates of evaporative loss (Figure 6.14).

7.4.5 DITCH WATER LEVELS, AGRICULTURE AND NATURE CONSERVATION

Regardless of evaporation and evapotranspiration, the key influence on raising ditch water levels on the Pevensey Levels are the wishes of local landowners, particularly those involved in agriculture. On the Pevensey Levels, although wetland restoration has been attempted using the integrative water level regimes associated with the WES (Section 2.8.1), prescribed water levels have had an impact on local farming businesses. Key areas of concern have been the effects of revised water level prescriptions on the inundation of gateways and field drains, and the increased risk of inundation and waterlogging during heavy rainfall due to already high water levels. In dry years, a key issue has been how to maintain wet fencing and irrigation, a problem that has gained increasing recognition as awareness of climate change has increased within the Pevensey Levels Study Group (Section 6.4.5).

Field-scale modelling studies have been employed to address all these issues. During field surveys on the SWT Reserve, a number of gateways were levelled relative to ditch water levels. These levels were implemented within PINHEAD to evaluate the effects of various sluice level management regimes on gateway inundation (Section 6.4.3.1). The suitability of ditch water level regimes for grazing has been highlighted in model simulations. Setting sluices to levels accordant with the requirements of graziers serves to eliminate the inundation of gateways and fields throughout the entire four-year period considered. In contrast, implementation of the WES results in water levels exceeding the mean gateway elevation in wetter years, such as 1997 and 1998, when model results indicate that gateways would be inundated for 15 and 33 days respectively. Implementation of ESA water level prescriptions results in an even greater increase in the frequency of gateway inundation. Model results show that gateways would be inundated for more than 75 days in wet years and more than 31 days in dry years if ESA water levels were implemented on the SWT Reserve (Table 6.14).

There is considerable evidence however that agricultural water level management to limit inundation has led to a considerable decline in the biodiversity value of the site (Sections 6.4.1 and 6.4.2). Conservationists have highlighted the progressive decline in the numbers of breeding and over-wintering waders and anatids on the wetland (Figure 2.15). This decline is similar to trends noted for other wet grasslands in the UK (Figure 1.3). These decreases are generally ascribed to the lower water levels required by agriculture and by the influence of pump drainage that limits inundation of field surfaces. The impacts of both practices on birds has been confirmed by the large negative exceedances beyond the requirements of birds associated with sluice settings for agriculture (Figure 6.5). However, water levels maintained for grazing have been found to be highly suitable for the rare plant species and the fen raft spider (Figure 6.7.a), helping to explain the current biodiversity status of the site. Model results do however confirm the large negative impacts that pump-drainage has on the ditch habitat, leading to large exceedances throughout the year below the requirements of key species (Figure 6.7.b). This, at least in part, may explain why rare plant species and populations of the Fen Raft spider are concentrated in the gravity drained area (Figures 2.14 and 2.16), the only part of the wetland outside the influence of the pumps.

7.4.6 THE IMPACTS OF CLIMATE CHANGE

Results presented in this thesis have highlighted the large influence of prevailing climatic conditions (the balance between rainfall and evaporation) on the wetland water balance (Section 3.7.1) and wetland stakeholders. Effects on stakeholders range from the influence of dry summers on the ability of farmers to provide sufficient irrigation for grass crops, to the difficulties in raising ditch water levels due to the importance of evaporation and evapotranspiration. Such difficulties are those which drive the need for flexible approaches to management, more specifically those related to water level control. Results presented in this thesis clearly highlight that sluice keepers and landowners must respond to weather conditions proactively throughout the year to ensure wet fences and irrigation are provided in the summer, and sufficient flood storage capacity is available during the winter months.

The precise degree of flexibility required is likely to increase under climate change scenarios. Given the considerable media coverage regarding global climate change in recent years, members of the Pevensey Levels Study Group have frequently shown interest in the potential effects of climatic change on wetland hydrology and management (Section 6.4.5). In particular, the drought years of 1995, 1996 and 2003 have raised awareness of water resource issues on the wetland, including the sustainability of existing and proposed water level management strategies. It is expected that climate change will have especially large impacts on water level management practices adopted during the summer months. This issue gains further importance given the current difficulties of satisfying stakeholder requirements during dry summers

(Section 6.4.4) where even retaining higher winter water levels has a limited effect on the drying of ditches during drought years.

Current estimates suggest that in the UK, climate change will result in higher winter rainfall and drier summers. The effects of such changes on wetland water levels have been discussed in detail in Section 6.4.5. Simulation of a range of climate change scenarios using PINHEAD indicate that the effects of climate change on wetland hydrology will be subtle, but significant. Although predicted increases in winter rainfall will lead to small increases in the frequency of inundation under certain sluice management scenarios, the greatest effects will be on summer water levels, where increased rates of evaporation and reductions in rainfall predicted by GCMs will result in mean annual water levels 0.04-0.07m lower than at present (Section 6.4.5). The most notable reductions are in the crucial early and mid-summer months, a period of importance to both nature conservation and agricultural stakeholders on the wetland. Of particular interest is the fact that for all years considered, any increases in water levels resulting from increased winter rainfall, or attained by raising sluice levels, are offset by the higher rates of evaporation that climate change models predict. Model results indicate that ditch water levels on the SWT Reserve are more sensitive to changes in evaporation and evapotranspiration during the summer than rainfall during winter (Figure 6.19).

The frequency of water levels less than 1.40m OD has been found to increase with climate change regardless of the scenario implemented or the water level management strategy adopted (Table 6.17). This is the case even though all the scenarios implemented are associated with increases in winter rainfall (see Table 6.17). Under water level management for grazing, model predictions indicate that the duration of 'dry' periods will increase in dry years. Changes in the frequency of water levels less than 1.40m OD are equivalent to increases of 5 days in 1995 and 14 days in 1996 (Figure 6.19.a). Similar impacts on the ability of wetland managers to satisfy the requirements of wetland biota are predicted. Results shown in Table 6.17 show that climate change predictions for 2080 will reduce the frequency of the highest water levels in all years. For each year in the period 1995-1998, changes in the frequency of inundation are equivalent to reductions of 12, 1, 6 and 12 days respectively relative to current climatic conditions.

7.5 Recommendations for management and future research

Results presented in this thesis demonstrate the need to partially reconsider hydrological management across the wetland. A key driver for change is the location of the wetland in one of driest regions of the UK, the excess of evaporation and evapotranspiration over rainfall during the summer, the potential effects of climate change, and the over-riding influence of all these factors on wetland stakeholders. Recommendations arising from the work presented in this thesis can be broadly subdivided into three key areas:

- Confirmation of key results presented in this thesis through continued data collection on the Pevensey Levels wetland;
- Enabling a more sustainable approach to hydrological management across the site;
- Suggestions for further technical / scientific work to be undertaken to understand the hydrology of the Pevensey Levels, wet grasslands and wetlands in general.

Confirmation of the results presented in this thesis will require continued monitoring of various aspects of the hydrology of the wetland, with specific targeting of key areas where data are currently lacking. Collection of these data will allow wetland managers to continue the pro-active approach to hydrological management on the site. Losses to the sea remain one of the hydrological processes that need to be considered in more detail. This is because, based on current methods for their calculation, they represent a significant outflow to the wetland on an annual basis. They also influence assessments of the sustainability of abstraction. A more flexible approach to management of losses to sea could provide a potential means of mitigating the effects on wetland stakeholders of current water scarcity during the summer months and securing an increased water resource for abstraction to supply the expanding towns of Eastbourne, Hailsham and Polegate. However, this will require continued collection of hydrological data describing the hydrology of the embanked channels on the wetland, including water levels, gate levels and volumes discharged by pumping stations at smaller time-steps than is currently available.

From the experiences obtained during this study, it is also possible to suggest a series of guidelines for the collection of hydrological data on the Pevensey Levels wetland. This may allow a more detailed and cost-effective method for water resource assessments in operational practice. Fundamentally these are:

1. **Storage of all available hydrological data in one location**, ideally in the Environment Agency Pevensey Office. These data should be routinely incorporated within a database such as the water balance model included as part of this thesis, or equivalent Environment Agency tool. Given the location of the wetland in one of the driest areas of England, the importance of the wetland in both nature conservation and agricultural terms and the increasing influence of legislative pressures on wetland managers, it is likely that other hydrological studies on the wetland will be necessary in the future and will benefit from the availability of such data.

2. **Collection of water level data and information describing pump functioning on a more temporally-intensive basis** using continuous loggers which are becoming increasingly affordable. The timing of data collection should be coincident for all pumping stations and channels on the wetland where monitoring is undertaken;

3. **Continued collection of water level data at the three key spatial scales of hydrological functioning** to enable wetland managers to identify the effects of management and prevailing climatic conditions on wetland stakeholders;

4. **Development of methods for the continuous estimation of losses to sea,** including the installation of continuous water level monitoring devices at all marine outfalls, recording of the elevation of main water gates on a regular basis and the deployment of flow measurement equipment to develop stage-discharge relationships for each outfall to estimate losses to sea on a continuous basis;

5. **An assessment of the volumetric contributions to field-scale ditches associated with feeding.** This may enable the identification of ways in which impacts on stakeholders during dry years can be mitigated. A particular focus should be an assessment of the timing of feeding, and the development of generic guidelines for feeding under different climatic conditions. This will necessarily require a detailed assessment of the hydrology of the channels that supply water for feeding (*e.g.* Wallers Haven),

6. **Use of water level data for the calculation of wetland evaporation and evapotranspiration** and implementation methods presented in this thesis, especially where wetland restoration strategies are in operation.

Results presented in this thesis also enable a series of proposals relating to the management of the site, including:

1. **Detailed surveys of target drainage system should be undertaken prior to raising ditch water levels or installing structures for water level control**. Such surveys should be an integral component of the implementation of wetland restoration strategies and should include the levels of gateways, existing structures, and field surfaces within the target area. In many wetland areas located in floodplain areas or prone to flooding, laser altimetry data (LIDAR) may be available (these data are available on the Pevensey Levels wetland for example) and can be used to estimate field surface elevations. A knowledge of site topography will enable the detailed design of any structures to be installed in areas of nature conservation interest to ensure that penning levels associated with wetland restoration objectives can be provided, and that in areas of agricultural importance, basic farming operations are not compromised.

2. **To limit areas of raised water levels across the wetland**. Areas where nature conservation value is currently greatest and/or the infrastructure of the drainage system allows flexible water level management of the site should be targeted. Water scarcity during the summer months and calculations of the demands imposed by revised water level management strategies wetland-wide has suggested that under current management practices there is insufficient water to provide blanket implementation of higher water levels across the site.

3. **A review of the method employed to estimate flows at Boreham Bridge**. Results indicate that the factor formula tends to over-estimate inflows from upstream catchments. A detailed comparison of data provided by the ultrasonic gauge at Boreham Bridge relative to flows estimated using the factor formula should be undertaken once a significant volume of data is available from the ultra-sonic gauge. This assessment has a bearing on abstraction from Boreham Bridge as the licence is subject to a flow condition based on the factor formula.

4. **Production of a hydrological plan for the management of the Wallers Haven**. The plan would provide a detailed evaluation of the hydrology of the channel, review management practices as a means of identifying opportunities to satisfy

water demand from both abstractors and wetland stakeholders, and 'fine-tune' management of feed sluices along the channel length. It would also consider the likely effects of climate change on future water resource demands on the channel. This will ensure that the demand for potable water supply is satisfied at present and in the future, and that impacts on the ability of wetland stakeholders to manage water levels within prescribed objectives is not compromised.

Studies undertaken in this thesis have also identified various aspects of the hydrological functioning of wet grassland that require further consideration. All have the ability to considerably influence management approaches in wetland areas. Based on modelling approaches implemented in this thesis, further studies are considered necessary with regards to:

1. **Development of methods for the estimation of sluice discharge.** Comparison between estimates of flow through penning board sluices and estimates provided by equations commonly employed for the estimation of weir discharge indicate the inadequacy of such methods for estimating flows through these structures. Results provided in this thesis also highlight the important role of aquatic vegetation in controlling flow and partially support observations of the ability of wetland channels to retard flood flows.

2. **Further studies regarding the dynamics of runoff generation in lowland/wetland areas.** The lack of a relationship between rainfall, runoff coefficients, and parameters commonly employed for the estimation of catchment responses to rainfall events (*e.g.* soil moisture conditions as employed in the FSR flood estimation approach) suggest that a more detailed understanding of the dynamics of runoff generation in lowland wet grassland is required. This is particularly applicable to the study of the effects of raising ditch water levels on wetland hydrology and the wetland water balance. One potential feedback mechanism is that raising ditch water levels will increase runoff volume during rainstorm events. However, to date, no methods have been identified in the literature to quantify this process. Such information is required to accurately consider the impacts of wetland restoration on wetland stakeholders and the sensitivity of wetland systems managed for nature conservation objectives to climate change.

These issues are also of significance in the context of flood generation and the provision of suitable levels of flood defence in restored wetland areas.

3. **Continued evaluation of the dynamics of evaporation and evapotranspiration in wetland areas**. The over-riding influence of evaporation and evapotranspiration in terms of the catchment-scale and field-scale water balance of the Pevensey Levels suggests that special attention should be given to the accuracy of estimates employed in operational practice. Results presented in this thesis have been confirmed at other locations (Gavin, 2000) and illustrate that wetland evapotranspiration can proceed at a rate greater than potential evapotranspiration, a process that is likely to involve complex feedback mechanisms between the wetness of the wetland surface, the proportional cover of open water, and the vegetation communities that different hydrological regimes encourage. Studies should consider a variety of different wetland habitat types.

4. **Development of generic approaches to quantify all components of the wetland water balance and guidelines to implement these in a holistic manner.** Throughout this thesis, the wetland water balance has emerged as a key concept within the study of wetland hydrology, management and the influence of these aspects on local stakeholders. RSPB *et al.* (1997) suggest a methodology which relies solely on rainfall, evaporation and soil moisture deficit data. The estimation of ditch and water table storage should also be considered, as storage is a key characteristic, and the *raison d'etre*, of most wetland environments (Hollis and Thompson, 1998). The wetland water balance should be quantified based on a conceptual model of the hydrological functioning of the site at a variety of spatial scales. Conceptual models of different wetland sites may be provided by the continued development and implementation of wetland classification schemes such as that presented by Gilvear and McInnes (1994) for wetlands in East Anglia. This approach classifies wetlands according to the relative importance of their inflows, outflows and sinks. In particular, the implementation of the wetland water balance approach will enable a more accurate calculation of water resource availability in wetland areas with regards to the requirements of nature conservation and agriculture, thus providing a more sustainable approach to wetland management, both locally and at the catchment scale.

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APPENDICES

APPENDIX 2.1. Special Protection Area citation for the Pevensey Levels.

Pevensey Levels

SPA/Ram Code	IBA Europe number
1208A	203

County/Region	District(s)	OS sheet(s)	Grid Reference(s)	Maps 19, 20
East Sussex	Rother, Wealden	199	TQ 6507	

Area (ha)	NNR	SPA	Ramsar
1000-9999	Part	Designated N Candidate Y	Designated N Candidate Y

An extensive area of grazing marsh, attracting important numbers of breeding and wintering waterfowl.

Site description

The Pevensey Levels are a large area of low-lying grazing marsh to the north-east of Eastbourne. Originally an area of intertidal mudflats, the Levels developed first to saltmarsh and then to freshwater marsh as a result of land-claim. The deposition of a shingle beach along the present coastline aided this process, and it now protects the Levels from sea water inundation. The maintenance of ditches helps to create a range of ditch types allowing a diverse floral and invertebrate community to become established. Several nationally scarce aquatic plant species are present, notably pondweed species. The main channels which carry water to the sea are less species-rich.

Most fields are of improved rye-grass leys with some creeping bent. A small area of shingle and intertidal mud and sand is included within the site.

The Pevensey Levels are of national importance for molluscs and aquatic beetles, including the rare great silver water beetle, Britain's largest water beetle. Over 15 species of dragonfly have been recorded from the Pevensey Levels including the nationally scarce hairy dragonfly and the variable damselfly.

Birds

The site supports an important assemblage of breeding bird species typical of lowland wet grassland. These include mute swan, mallard, lapwing, snipe, redshank, yellow wagtail, sedge warbler, reed warbler and reed bunting.

In winter the area is notable for supporting large numbers of lapwing and snipe.

Conservation issues

There have been massive losses of lowland wet grassland habitat in Britain in recent decades as a result of drainage and agricultural intensification. Remaining areas need strong protection from such damage. Much of the ornithological interest of the Pevensey Levels has been damaged in recent years due to increased drainage. Threats include further conversion to arable, road improvements, and lack of traditional management.

Further reading

The Sussex Ornithological Society. Sussex Bird Report.

APPENDIX 2.2. SSSI citation for the Pevensey Levels wetland.

COUNTY: EAST SUSSEX **SITE NAME:** PEVENSEY LEVELS

DISTRICT: WEALDEN; ROTHER

Status: Site of Special Scientific Interest notified under Section 28 of the Wildlife and CountrysideAct 1981. Part of this site has been designated a National Nature Reserve under Section 16 of the National Parks and Access to the Countryside Act 1949.

Local Planning Authority: WEALDEN DISTRICT COUNCIL; ROTHER DISTRICT COUNCIL

National Grid Ref: TQ 650 070 **Area:** 3501.0ha (8650.9 acres)

Ordnance Survey Sheets 1:50,000: 199 **1:10,000:** TO 60 SW; TQ 60 SE; TO 60 NW; TO 60 NE; TO 61 SW; TO 61 SE; TO 70 NW

Date notified: (under 1949 Act): 1977 **Date notified** (under 1981 Act): 1990
Other Information: This site is listed in *A Nature Conservation Review*. Part is a National Nature Reserve.

Reasons for Notification

Pevensey Levels is a large area of low-lying grazing meadows intersected by a complex system of ditches which show a wide variety of form and species composition and support important communities of wetland flora and fauna. The site supports one nationally rare and several nationally scarce aquatic plants and many nationally rare invertebrates. Ornithologically, the site is of national importance as the number of wintering lapwings has regularly exceeded 1 % of the total British population in recent years.

Geologically, the Levels are located where impervious Weald Clay reaching the coast has been overlain by superficial alluvial deposits. In places, however, the Weald Clay itself forms outcrops as at Horse Eye and Tunbridge Wells Sands reach the surface occasionally, as on part of Hooe Level. Once an area of intertidal mudflats, the Levels have developed in turn to salt marsh and fresh water marsh. This process has been aided by the deposition of shingle beach deposits, by the process of longshore drift, along the present coastline. This shingle ridge now protects the Levels from sea water inundation, since most of the site lies below the level of highest tide. Past intersections of the marshes by a series of ditches has created the present day area of rich grazing meadows.

The ditch system facilitates removal of surface water to enable successful stock grazing, at the same time acting as a network of "wet fences" and as a source of stock drinking water. Maintenance of the ditches is necessary to continue efficient execution of these functions and also creates a wide variety of ditch types from intensively or recently dredged ditches to neglected ones. In this way a wide variety of floral conditions prevail and the specific requirements of certain invertebrates are always catered for. Following the dredging of a clogged ditch a distinct successional pattern occurs. First, floating and submerged aquatic plants such as duckweeds *Lemna* spp., pondweeds *Potamogeton* spp or water fern *Azolla* spp colonise. These are followed by larger floating or emergent plants such as frog-bit *Hydrocharis morsus-ranae,* bur-reed *Sparganium erectum* and arrow-head *Sagittaria sagittifolia.* Finally, common reed *Phragmites australis* becomes dominant at the expense of most other species. If left undredged the ditches may dry up and become scrubbed over with drastic effects on plant and animal diversity.

The most species-rich ditches show a varied structure and a good mixture of both open water and emergent species. The broad-leaved pondweed *Potamogeton natans* and frog-bit are abundant, whilst the nationally rare* sharp-leaved pondweed *Potamogeton acutifolius* (RDB:*** Vulnerable) is of particular importance. Other open water species include ivy-leaved duckweed *Lemna trisulca* and the nationally scarce** water soldier *Stratiotes aloides* and flat-stalked pondweed *Potamogeton friesii.* Numerous other pondweeds are found here including shining pondweed *Potamogeton lucens,* curled pondweed *Potamogeton crispus* and blunt-leaved pondweed *Potamogeton obtusifolius.* Emergents of interest include the. nationally scarce greater water-parsnip *Sium latifolium* and river-dropwort *Oenanthe fluviatilis.* These very species-rich ditches are largely confined to gravity-drained areas within the site.

The main arterial channels, which carry drainage water from the Levels to the sea, are generally poor in vegetation, both in number of species and cover. Submerged and floating species such as common duckweed *Lemna minor* and greater duckweed *Lemna polyrhiza* predominate with the nationally scarce spineless homwort *Ceratophyllum submersum* and the nationally scarce pondweed *Potamogeton trichoides* also present. Ditches surrounding and within arable areas support relatively few open water species and tend to be characterised by the presence of water plantain *Alisma plantago-aquatica* and bur-reed. They are often fringed with hard rush *Juncus inflexus* and jointed rush *Juncus articulatus.*

Rich bankside floras support the nationally scarce marshmallow *Althaea officinalis,* ragged robin *Lychnis flos-cuculi,* water mint *Mentha aquatica* and cuckoo flower *Cardamine pratensis.* Most of the fields are improved rye grass *Lolium perenne* leys with occasional creeping bent *Agrostis stolonifera.*

Woodland dividing the main Pevensey to Middle Bridge road from the old road parallel to it is 0 dominated by mature crack willow *Salix fragilis* with hawthorn *Crataegus omonogyna* and elder *Sambucus. nigra.* Closed canopies have a sparse ground cover of ground ivy *Glechoma0 hederacea* and nettle *Urtica dioica.* This area is of importance for moths.

An area of shingle and intertidal muds and sands is included within the site. Although the shingle is largely bereft of vegetation, yellow horned-poppy *Glaucium flavum,* sea campion *Silene maritima* and the nationally scarce sea kale *Crambe maritima* do occur; there is also a record for pyrimidal orchid *Anacamptis pyramidalis.*

The site supports outstanding invertebrate populations and is a top national site for Mollusca and aquatic Coleoptera. Indeed, the site is perhaps the best in Britain for freshwater Mollusc fauna. A ramshorn snail *Segmentina nitida* (RDB: Endangered), is found in well-oxygenated drains with lush vegetation. Particularly abundant and widespread on this site is an aquatic snail *Valvata macrostoma* (RDB: Vulnerable). Of the many species of water beetle recorded at the site, the most interesting are confined to the ditches in areas of permanent pasture. Of particular interest is Britain's largest water beetle, the great silver water beetle *Hydrophilus piceus* (RDB: Rare) which is found only on grazed levels in the southern part of Britain. Also of importance is *Bagous puncticollis* (RDB: Endangered), found on Horse Eye Level and several nationally rare water beetles such as the small reddish-brown *Hydrovatus clypealis* (RDB: Rare) confined to the coast of southern England.

Over fifteen species of dragonfly (Odonata) have been recorded including the nationally scarce hairy dragonfly *Brachytron pratense* and variable damselfly *Coenagrion pulchellum.* Survey has also revealed Britain's only known location of *Placobdella costata* (provisional ROB), a large leech which feeds on the blood of vertebrates. One of Britain's largest spiders *Dolomedes plantarius* (ROB: Endangered) has also been recorded. The site is of national importance for its wintering lapwing *Vanellus vanellus* which exceed 1 % of the total British population. The numbers of snipe *Gallinago gallinago* may also be of national importance but exact data relating to the country's wintering population is as yet unavailable. Wintering golden plover *Pluvialis apricaria* are of local significance and in some years are of national importance. Sedge warblers *Acrocephalus schoenobaenus* and reed warblers *Acrocephalus scirpaceus,* which nest in scrub close to water and reeds in the ditches respectively, breed in numbers of local significance. The site also supports about one fifth of the breeding wagtails *Motacilla flava* in Sussex.

NOTE

*Nationally Rare Occurs in less than 15 of 10 X 10km squares in Britain

**Nationally Scarce Occurs in 15-100 of 10 X 10km squares in Britain

***ROB Nationally rare species are fisted in the relevant Red Data Book (RDB), two of which have been published: "British Red Data Book 1: Vascular Plants" and "British Red Data Book 2: Insects". The three RDB categories: Rare, Vulnerable and Endangered indicate increasing degrees of extinction in Britain.

APPENDIX 2.3. Ramsar citation for the Pevensey Levels wetland.

Ramsar Convention on Wetlands of International Importance Especially as Waterfowl Habitat

PEVENSEY LEVELS (EAST SUSSEX)

Pevensey Levels proposed Ramsar site represents one of the largest and least fragmented lowland wet grassland systems in south-east England. The low-lying grazing meadows are intersected by a complex system of ditches which support a variety of important wetland communities, including nationally rare and scarce aquatic plants and invertebrates. The site also supports a notable assemblage of breeding and wintering waterfowl. The boundary of the proposed Ramsar site follows that of the Site of Special Scientific Interest, notified in 1990 under the Wildlife and Countryside Act, 1981. The site qualifies under Criterion 2a of the Ramsar Convention by supporting an outstanding assemblage of wetland plants and invertebrates including many Red Data Book (RDB) species. The following RDB invertebrates have been recorded: the ramshorn snails *Segmentina nitida* (RDB: Endangered) and *Anisus vorticulus* (RDB: Vulnerable), an aquatic snail *Valvata macrostoma* (RDB: Vulnerable), the great silver water beetle *Hydrophilus piceus* (RDB: Rare), the waterbeetles *Graphoderus cinereus, Hydraena pulchella, Hydrochus elongatus, H. ignicollis, Ochthebius exaratus* and *O. pusillus* (RDB: Rare), a whirligig beetle *Gyrinus suffriani* (RDB: Rare), a beetle *Telmatophilus brevicollis* (RDB: Rare), a weevil *Bagous puncticollis* (RDB: Endangered), a bug *Hydrometra gracilenta* (provisional RDB: Rare), the fen raft spider *Dolomedes plantarius* (RDB: Endangered), a horsefly *Atylotus rusticus* (RDB: Endangered), a soldier fly *Odontomyia omata* (RDB: Vulnerable), the snail killing flies *Pherbellia argyra* and *Psacadina zemyi* (RDB: Vulnerable), the craneflies *Limophila pictipennis* (provisional RDB: Vulnerable) and *Tipula marginata* (RDB: Rare), and the leech *Placobdella costata* (provisional RDB) at its only known location in Britain. The sharp-leaved pondweed *Potamogeton acutifolius* (RDB: Vulnerable) occurs in species rich ditches.

The site also qualifies under Criterion 2b of the Convention, as it is of special value for maintaining the genetic and ecological diversity of the region. It is probably the best site in Britain for freshwater Molluscs, one of the five best sites for aquatic

Coleoptera and supports an outstanding assemblage of dragonflies (Odonata). Of 160 plants in Britain which can be described as aquatic, about 110 (68%) are found on Pevensey Levels. The site supports an important assemblage of breeding wetland birds typical of lowland wet grassland, including lapwing *Vanellus vanellus*, snipe *Gallinago gallinago*, redshank *Tringa totanus* and yellow wagtail *Motacilla flava*. In winter the site supports notable populations of snipe, lapwing and golden plover *Pluvialis appricaria*.

June 1994

APPENDIX 4.1. Notation for Table 4.2.

a is constant of proportionality that is established locally (=1-1.3), but commonly 1.26 (Shuttleworth (1978).

C is an adjustment factor which depends on minimum relative humidity, sunshine hours and daytime wind estimates.

C_p specific heat of moist air.

e_s-e is vapour pressure deficit (kPa).

G is soil heat flux.

P is the mean daily percentage of total annual daytime hours for a given month and latitude.

R_s solar radiation (mm d^{-1}).

T is the mean daily temperature over the month considered (^{o}C).

W weighting factor which depends on *T* and altitude.

ρ is atmospheric density (kg m^{-3}).

Δ is the change of saturated vapour pressure with temperature (kPa $°C^{-1}$).

γ is the psychrometric constant (kPa $°C^{-1}$).