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Integrated catchment scale model of a lowland eutrophic lake and river system: Norfolk, UK

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ABSTRACT

Shallow lakes are ecologically and economically important; many users are interested in methods to assess their response to restoration measures and in tools to predict the impact of specific measures. These users include: local and governmental authorities, private companies or nature conservation organisations.

This research is centred on the Broads. The Broads are shallow, eutrophic lakes, probably the result of medieval peat workings, concentrated in the Ant, Bure, Thurne and Yare river valleys. These man made lakes and their surroundings are unique in Europe in terms of both ecology and landscape, forming one of the few remaining large areas of lowland river grassland in the UK.

A catchment scale model, SWAT, has been used to model past and future land use and climate scenarios for river basins supplying water and nutrients to the Broads. SWAT is a comprehensive model that requires a diversity of information including climate, topography, soil, land use, agricultural practices, water abstraction and discharge data.

Future scenarios run with SWAT suggest that increases in rainfall and temperature through climate change and changed land use increase nutrient and sediment yields and runoff. Future scenarios therefore suggest increased eutrophication problems for both the rivers and Broads within the study area and an increase in the already high risk of ecological failure to the Broads.

Various management scenarios based on erosion control measures were designed to alleviate nutrient and sediment yields and increased run-off to the system. SWAT modelling showed the best-case future scenario in terms of land management was to convert the area to grassland. Where land is still used for agriculture erosion control, measures such as cover crops and conservation tillage should be employed.

Overall, the work has increased the understanding of water quality, water movement, nutrient and sediment dynamics and agricultural management practices within the study area. The environmental implications of different future scenarios and erosion control measures on the ecology of the Broads provide a basis for management of the area.

ACKNOWLEDGEMENTS

Catchment modelling is 25% model, 50% the modeller
and 25% good luck!!! (http://euroharp.org)

I would like to express my sincere thanks and gratitude to the many people who have participated in various ways to ensure my project is completed. They have provided me with encouragement, sound advice, good teaching and lots of good ideas.

My parents have been an inspiration throughout my life. They have always supported my dreams and aspirations. I'd like to thank them for who and what they are, and all they have done for me.

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8 Ugly; Beth Orton; Blur; Bob Marley; Crowded House; Green Day; Martha Wainwright; Teddy Thompson; Beatles; Oasis; Norah Jones; Kaiser Chiefs; Franz Ferdinand; Ocean Colour Scene; Paul Weller; R.E.M; Red Hot Chilli Peppers, Sterophonics and U2.

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%	percent
<	less than
>	greater than
√	square root
£	sterling pound
°C	degrees Celsius
0361	Sandwich soil
0541s	Wick soil
0711v	Gresham soil
0871c	Hanworth soil
1022b	Altcar 2 soil
A	Conversion factor
ALPHA_BF	base flow alpha factor
a_s	regression constant, expressing the fraction of extraterrestrial radiation reaching the earth on overcast days
a_s+b_s	fraction of extraterrestrial radiation reaching the earth on clear days (n = N).
BIOMIX	bio-mixing efficiency
cm	centimeters
CN	curve number
CO_2	carbon dioxide
c_p	specific heat at constant pressure (MJ kg^{-1} $^oC^{-1}$)
E	depth rate evaporation (mm d^{-1})
e	accepted error measured as a proportion of the standard deviation
e_d	vapour pressure (kPa)
Elv	elevation above sea level (m)
E_{NS}	Nash and Sutcliffe efficiency
E_o	potential evapotranspiration (mm d^{-1})
e^o_z	saturation vapor pressure of air at height *z* (kPa)
ESCO	soil evaporation compensation
es_{db}	saturation vapour pressure (kPa)
es_{wb}	saturation vapour pressure (kPa)
ET	evapotranspiration
et al.	and others
ez	water vapor pressure of air at height *z* (kPa)
f_{cl-si}	soil erodibility factor soils with high clay to silt ratios
f_{csand}	soil erodibility factor with high clay to sand ratios
f_{hisand}	soil erodibility factor for soils with extremely high sand contents
f_{orgc}	soil erodibility factor for soils with high organic carbon content
FRT_LY1	fertiliser application rates
G	heat flux density to the ground (MJ m^{-2} d^{-1})
gm^2 yr^{-1}	grams squared per year
GS	global sustainability
GWHT	groundwater height (m)
GWQMN	minimum depth of water in soil for base flow to occur (m)
GWREVAP	groundwater re-evaporation coefficient
H_0	extraterrestrial radiation (MJ m^{-2} d^{-1})
ha	hectares

H_{net}	net radiation (MJ m^{-2} d^{-1})
K	hydraulic conductivity (mm h^{-1})
kg	kilogram
km	kilometre
km^2	kilometre squared
kPa	kilo Pascal's
K_{SAT}	saturated hydraulic conductivity
K_{sat}	saturated hydraulic conductivity (mm h^{-1})
kWh	kilowatts per hour
mm hr^{-1}	millimetres per hour
m $year^{-1}$	metres per year
m	meter
m^2 yr^{-1}	metres squared per year
m^3 s^{-1}	metres cubed per second
m_c	soil clay content (%)
mg l^{-1}	milligrams per litre
MJ m^{-2} d^{-1}	mega joules per meter squared per day
MJ	mega joules
mm	millimetres
m_s	soil sand content (%)
m_{silt}	soil silt content (%)
MUSLE	modified universal soil loss equation
n	actual duration of sunshine (hour)
N	maximum possible duration of sunshine or daylight hours (hour)
N	nitrogen
n/N	relative sunshine duration
NH_3	ammonia
NH_4	ammonium
NO_2	nitrite
NO_3	nitrate
NPERCO	nitrogen percolation coefficient
orgC	soil organic carbon content (%)
ORGN	organic nitrogen
ORGP	organic phosphorus
P	air pressure (kPa)
P	phosphorus
PET	potential evapotranspiration
PHOSKD	phosphorus soil partitioning coefficient
PPERCO	phosphorus percolation coefficient
ppm	parts per million
R^2	fraction of the total squared error
r_a	diffusion resistance of the air layer (aerodynamic resistance) (s m^{-1}).
R_a	extraterrestrial radiation (MJ m^{-2} day^{-1})
r_c	plant canopy resistance (s m^{-1})
RE	regional enterprise
REVAPMN	minimum depth of water in shallow aquifer for re-evaporation to occur (m)
R_s	solar or shortwave radiation (MJ m^{-2} day^{-1})
RSDCO	crop residue factor
s	the sample size

SLOPE	slope of HRU (m)
SLSUBBSN	USLE slope length (m)
SOL_MINP	concentration of soluble phosphorus in soils
SOL_NO3	concentration of nitrogen in soils
SOL_ORGN	concentration of organic nitrogen in soils
SOL_ORGP	concentration of organic phosphorus in soils
SOLP	soluble phosphorus
$t\ ha^{-1}$	tonnes per hectare
T_{av}	mean air temperature for a given day (°C)
T_{cal}	calculated student T test value
T_{db}	dry bulb temperature (°C)
Tmn	minimum air temperature for a given day (°C)
Tmx	maximum air temperature for a given day (°C)
T_{tab}	tabulated student T test value
T_{wb}	wet bulb temperature (°C)
USLE_C	crop practice factor
USLE_P	USLE crop management factor
z	degree of confidence for a z test
$\mu g\ l^{-1}$	micrograms per litre
µg	micrograms
ρ_{air}	air density ($kg\ m^{-3}$)
Σ	sum
Δ	slope of the saturation vapor pressure-temperature curve
γ	psychrometric constant ($kPa\ ^{o}C^{-1}$)
λ	latent heat of vaporization ($MJ\ kg^{-1}$)
λE	latent heat flux density ($MJ\ m^{-2}\ d^{-1}$)

ABBREVIATIONS AND ACROYNMS

ACCELERATES	Assessing Climate Change Effects on Land use and Ecosystems
ADAS	Agricultural Development and Advisory Service
ANSWERS	Areal Non-point Source Watershed Environmental Response Simulation
AOGCM	Atmosphere-Ocean General Circulation Model
AONB	Area of outstanding Natural Beauty
ASTM	American Society for Testing and Materials
AWC	Available Water Capacity
BADC	British Atmospheric Data Centre
BFI	Base Flow Index
CAP	Common Agricultural Policy
CEH	Centre of Ecology and Hydrology
CGCM1	Canadian Global Coupled Model 1
CN	Curve Number
CREAMS	Chemicals, Runoff and Erosion from Agricultural Management System
CropWat	Crop water requirements software package
cSAC	candidate for Special Area of Conservation
CSIRO-MK2	Commonwealth Scientific and Industrial Research Organisation
DEFRA	Department for Environment, Food and Rural Affairs
DEM	Digital Elevation Model
DETR	Department of the Environment, Transport and Regions
DHI	Danish Institute of Hydrology
DO	dissolved oxygen
Dr.	Doctor
DRMS	Daily Root Mean Squared
DTI	Department of Trade and Industry
DWF	Dry Weather Flow
E	Estimated
EA	Environment Agency
EC	European Community
ECHAM4	European Meteorological Centre Hamburg
EDL	Edinburgh Data Library
EEC	European Economic Community
NNR	National Nature Reserve
ESA	Environmentally Sensitive Area
ET	Evapotranspiration
EU	European Union
EUROHARP	European Harmonised Procedures for quantification of nutrient losses from diffuse sources
FAO	Food and Agriculture Organisation of the United Nations
Fig.	figure
FWR	Foundation for Water Research
GCM	Global Circulation Model
GLEAMS	Groundwater Loading Effects of Agricultural management Systems
GQA	General Quality Assessment
GS	Global Sustainability

HadCM2	Hadley Centre Coupled Model 2
HadCM3	Hadley Centre Coupled Model 3
HOST	Hydrology of Soil Type
HRU	Hydrological Response Unit
HSPF	Hydrologic Simulation Program-Fortran
ICM	Integrated Catchment Management
IDB	Internal Drainage Board
IH	Institute of Hydrology
INCA	Integrated Nitrogen in Catchments Model
INCA-P	The integrated Catchments Model of Phosphorus dynamics
IPCC	Intergovernmental Panel on Climate Change
LAI	Leaf Area Index
LandIS	Land Information System
LARS-WG	Weather Generator
M	Measured
MAFF	Ministry of Agriculture, Fisheries and Food
MIKE-SHE	Generalised River Modelling Package-Systeme Hydroloque Europeen
MONARCH	Modelling Natural Resource Responses to Climate Change
MORECS	Meteorological Office Rainfall and Evapotranspiration Calculation System
MUSLE	Modified Universal Soil Loss Equation
N	Nitrogen
NGR	National Grid Reference
NRCS	Natural Resources Conservation Service (United States)
NSRI	National Soil Resources Institute
NVZ	Nitrate Vulnerable Zone
Obs.	Observed
OD	Ordnance Datum
osr	oilseed rape
p	peas
P	Phosphorus
PBIAS	Percentage Bias
PET	Potential Evapotranspiration
Ph.D	Doctor of Philosophy
PLUARG	Pollution Land Use Activities Reference Group
pts	potatoes
RE	Regional Enterprise
RegIS	Regional Climate Change Impact and Response Studies
sa	set aside
SAC	Special Area of Conservation
sb	spring barley
sbt	sugar beet
SCS	Soil Conservation Service
Sim.	Simulated
SMD	Soil Moisture Deficit
SPA	Special Protection Area
ss	suspended sediment
SSSI	Site of Special Scientific Interest
STW	Sewage Treatment Works

SW	soil water
SWAT	Soil Water Assessment Tool
SWATCATCH	Surface Water Attenuation Catchment model
TERRACE	Terrestrial Runoff Modelling for Risk Assessment of Chemical Exposure
TON	Total Oxidised Nitrogen
UKCIP	United Kingdom Climate Impacts Programme
USDA	United States Department of Agriculture
USGS	United States Geological Survey
USLE	Universal Soil Loss Equation
wb	winter barley
wbn	winter field beans
WFD	Water Framework Directive
WUS	consumptive water use file
ww	winter wheat

Chapter One Introduction

1.0 Introduction

Throughout the world shallow lakes are more abundant and important to people and wildlife than deep lakes. Despite this, knowledge about them has not developed as rapidly as knowledge of deep lakes, but progress is now being made (Madgwick, 1999). They have a complex ecological structure and support the bulk of the biodiversity associated with freshwaters (Moss, 1998).

Shallow lakes are ecologically and economically important. Many users are interested in methods to assess their response to restoration measures and in tools to predict the impact of specific measures (Moss *et al.*, 1997). These users include: local and governmental authorities, private companies or nature conservation organisations.

Within the UK there are three lake districts. The English Lake District in Cumbria is well known, the second lake district, the North West Midland Meres is the least known. The third is a set of around 50 very small lakes linked by rivers called the Broads; on a world scale these lakes are tiny (Moss, 2001). However the area forms a network of wetlands that is unique in Europe in terms of both ecology and landscape, forming one of the few remaining large areas of lowland river grassland in Britain.

The 'Broads' are shallow lakes, probably the result of medieval peat workings (Lambert & Jennings, 1960). They are concentrated in the Ant, Bure, Thurne and Yare River valleys often fringed by fen and reedbeds with associated areas of carr woodland. In addition, a long history of settlement has left a legacy of historical and archaeological features. The Broadland area is rich in wildlife; the mosaic of wetland habitats in the Environmentally Sensitive Area (ESA) supports many rare and interesting species of plant, invertebrate and bird life.

1.1 Background

The entire Upper Thurne forms part of the Broads Special Area of Conservation (SAC) and is therefore an extremely important part of the Broads National Park in terms of biodiversity, recreation and agricultural use. In particular, Hickling Broad is the largest of all the Broads (116 ha) and is the richest site in the UK for the algal group of *charophytes*. *Charophytes* require clear water and good water quality to thrive. This more ecologically desirable, clear water, plant dominated system occurred in the Broadlands until the 1960's, but now only occurs in 3 out of the 41 broads in the national park. Hickling Broad also supports important populations of overwintering waterfowl and is renowned as a venue for boating.

In recent years Hickling Broad has been at the centre of scientific and management attention. In 1998 aquatic plants, particularly the intermediate stonewort (*Chara intermedia)*, multiplied and the water became clear for the first time in decades (Broads Authority, 1999). However in the winter of 1999/2000 the *Chara* lawns died back. Since then the broad has been in an unfavourable ecological condition, with turbid water and *Chara intermedia* confined to certain refugial areas.

In 2000 the *Chara* grew much less and there was a small resurgence in growth of *Prymnesium parvum*, which had caused fish kills in the early 1970's (Moss, 2001). Heavy grazing by coot during autumn 1999 and their consequent droppings is thought to have increased nutrient concentrations and resulted in algal blooms. In addition to a decrease in light penetration to Hickling Broad higher water levels added to the problem in 1999/2000 (Harris, 2001).

This sequence of events has led the Broads Authority to identify gaps in current knowledge and to develop clear research goals. They recognise the incomplete understanding of the system and wider catchment, particularly with respect to nutrient cycling. Therefore the Authority's overall aim is to develop a sound understanding of the functioning of the Upper Thurne system for future management purposes.

An understanding of the movements of water and nutrients within the Thurne and its wider catchment area (Bure and Ant) is required. Such an understanding will allow identification of the spatially and temporally varying sources of water and contaminants to the system. Furthermore evaluation through computer modelling of proposed changes in water and or nutrient management before they are implemented.

To meet these requirement the Broads Authority are supporting three PhD's in this area. To look at the river hydrodynamics, groundwater management and catchment scale modelling of diffuse pollutants. Together it is hoped they will be able to address the legal requirement of sustainable management of Hickling Broad, given by the Water Framework Directive (Directive 2000/60/EC).

This Directive requires the European member states to ensure that heavily modified water bodies (i.e. the Broads) reach 'good ecological potential' (DEFRA, 2004a). To achieve this in the Broads wiser management of the entire catchment as well as a cluster of local solutions will be required (Moss, 2001).

1.2 Aims and Objectives

Within this wider research topic the aim of this PhD 'An integrated catchment scale model of a lowland eutrophic lake and river system: Norfolk, UK' is:

To identify the effect of future climate and land use changes on flow patterns and nutrient loads in the Upper Thurne, Ant and Bure system.

To achieve this the following objectives will need to be addressed:

- To analyse current hydrological functioning of the catchments
- To apply a catchment model to current situations in the three areas
- To define likely future climate and land use scenarios
- To use the catchment model to quantify the impacts of future scenarios on flow and nutrient dynamics

The Upper Thurne is a very fragile ecosystem with a number of pressures on the system. These pressures are key to the more ecologically desirable clear water, plant dominated system that characterised the Broadlands until the 1960's. The physical structure and aquatic flora of the river ecosystem has been destroyed by the damage boats have done in the past, although boating activities are better managed now. The current water quality of the system is not compatible with diverse biological plant communities that are remembered from 50 years ago or more. There are over 200 sewage treatment works in the Broads. Many of which serve less than 1000 people and therefore do not fall under the Urban Waste Water Treatment legislation requiring works to undertake phosphorus stripping. There is still a great quantity of phosphorus being released into the Broadland system from sewage effluent, but legislation exists, were it to be used, to eliminate this source almost entirely.

Diffuse sources of nutrients from cultivated land, stock wastes, sediments and agricultural drainage systems are less easy to control but do however need to be addressed. Until water quality can be improved on a catchment wide-basis it is not worth attempting restoration measures such as the use of bio-manipulation in the Broads.

This thesis presents work carried out to address the aims and objectives specified above. Chapter 2 gives background information for the research with a review of understanding of nutrient sources, a review of catchment modelling and management and previous research related to this work. An overview of the study area in terms of available data for modelling purposes is given in chapter 3. An investigation into the sources and dynamics of nutrients in the study area system is presented in chapter 4. Chapter 5 outlines the catchment model build process using the Soil Water Assessment Tool (SWAT) for the first model and chapter 6 discusses the calibration and validation of this model. From the calibration and validation results of the first SWAT model the building of the second SWAT model to be undertaken in this research is explained in chapter 7. The definition of possible future scenarios with their impacts on nutrient loading, together with methods to minimise nutrient loads to the study area are given in chapter 8. Chapter 9 is for general discussion and conclusions.

Chapter Two **Literature Review**

2.0 Eutrophication

Nutrient enrichment (eutrophication) ranks as probably the most pervasive water quality problem on a global scale, potentially affecting all water bodies from rivers and lakes to estuarine and marine. The Urban Waste Water Treatment Directive defines eutrophication as:

'the enrichment of water by nutrients especially compounds of nitrogen and/or phosphorus compounds, causing an accelerated growth of algae and higher forms of plant life to produce an undesirable disturbance to the balance of organisms present in the water and to the quality of the water concerned' (91/271/EEC).

Phosphorus and nitrogen are the nutrients of most concern because of their primary role as potential growth limiting factors for phytoplankton (algae) and aquatic plants in water bodies (Novotny & Olem, 1994). In natural systems, nutrients are commonly derived from weathering and leaching from rocks and soils. However, nutrient inputs to aquatic ecosystems can be significantly accelerated by human activities, resulting in nutrient enrichment (Ostry, 1982). This represents anthropogenic eutrophication, which is in contrast to the natural ontogenic eutrophication process observed in water bodies.

Nutrient sources can be broadly segregated into two categories: readily identifiable point sources (such as sewage effluents) and diffuse sources (such as the run-off from agricultural land), with the relative contribution of each varying between river basins (Ongley, 1996).

For nitrogen, inputs to fresh waters in Europe come principally from diffuse sources, particularly agriculture, although point sources (usually urban wastewater) also contribute significantly in many regions. In England and Wales 70% of the total input of nitrogen to inland surface waters is estimated to come from diffuse sources (agriculture, precipitation and urban run-off, in order of decreasing importance). The remaining 30% comes from sewage effluent and industrial discharges (Morse *et al.*, 1993). For phosphorus, the actual contributions in any given river basin will depend on the nature of the river basin and the human activities within it (Heathwaite & Johnes, 1996).

In the past, point source pollution, for example wastewater treatment plant effluents, often formed the major nutrient inputs to water bodies. They are most obvious and easily identifiable sources (Ryding, 1986). Eutrophication control measures, including phosphorus removal from effluents, were often directed at such effluents. Over the last twenty years, legislation and regulation have become more effective at tackling point sources of pollution to ground and surface waters (DEFRA, 2002b). One such regulation is the Urban Waste Water Treatment Directive (91/271/EEC). This regulates the phosphorus content of discharges but only for larger sewage treatment works (STW's) where the discharge is to a 'sensitive area', as defined in the directive.

Data from the Environment Agency (EA) has shown that since sewage treatment work improvements have been brought on-line there has been a steady improvement in water quality. Nevertheless, rivers over large areas of England and Wales continue to show, high, very high or excessively high phosphorus levels (DEFRA, 2003). This classification is derived from the Environment Agency's General Quality Assessment Scheme (GQA) (see Table 2.0.1). The scheme provides a way of comparing river quality from one river to another and for looking at changes through time.

Table 2.0.1: GQA classification for phosphate

Classification for Phosphate	Grade limit (mg P l^{-1}) Average	Description
1	0.02	Very low
2	0.06	Low
3	0.1	Moderate
4	0.2	High
5	1.0	Very high
6	> 1.0	Excessively high

The EU Water Framework Directive (Directive 2000/60/EC) is the most significant piece of European water legislation to be produced for over twenty years. The

Directive, which will eventually replace a number of earlier ‘water’ directives, seeks to address water policy in a coherent, holistic and sustainable way for all waters. It requires all member states to achieve at least ‘good’ water quality status for all surface and ground water bodies by 2015. For surface waters ‘good’ is defined by both chemical and ecological parameters and requires a standard only slightly below that of a water body showing no effects resulting from activities of mankind i.e. in a pristine state (Chave, 2001). Table 2.0.2 gives an example of the EA classification of water bodies at risk of non-compliance with the Water framework Directive.

Table 2.0.2: Risk Category of lakes in Norfolk

Site Name	Point source pollution	Diffuse source pollution	Water abstraction and flow regulation	Physical or 'morphological' alteration	Alien species
Hickling Broad or Heigham Sound	Not at risk	At risk	Not at risk	Probably not at risk	Probably not at risk
Horsey Mere	Probably not at risk	Probably not at risk	Not at risk	Probably not at risk	Probably not at risk
Barton Broad	At risk	At risk	Not at risk	Probably not at risk	Probably at risk
Martham Broad or Martham Broad (North and South)	Probably not at risk	Probably not at risk	Not at risk	Probably not at risk	Probably at risk

It is apparent that the point source controls are not universally successful in eliminating anthropogenic eutrophication of lakes and reservoirs (Ryding & Thornton 1999). A major component of the nutrient budget of a water body has been found to arise from non point sources within a river basin. Significant non-point nutrient sources include agriculture and urban run-off, intensive livestock activities and atmospheric deposition (Vollenweider & Kerekes, 1982b). Because sources are diffuse, this pollution is difficult to measure and regulate. Ironically, due to intense efforts to increase the fertility of the land, waters have problems of excess fertility that impair water supply and create the need for costly remediation (Vollenweider, 1981).

Eutrophication can have both temporary and long-term effects on aquatic ecosystems (Anderson, 1995). Large fluctuations in dissolved oxygen concentrations can occur between day and night. Low oxygen levels, the result of plant respiration, may lead to the death of invertebrates and fish. This process can be compounded when algal blooms, through their decay, further reduce the oxygen content of water. The growth or decay of benthic (bottom-dwelling) mats of macro-algae can also lead to the deoxygenation of sediments (Cooke *et al.,* 1993).

Certain algal species, particularly freshwater blue-green algae and marine dinoflagellates can produce toxins, which may seriously affect the health of mammals (including humans), fish and birds. This occurs either through the food chain or through contact with, or ingestion of, the algae. Algal species also cause fish deaths, for example by physically clogging or damaging gills, causing asphyxiation. Eutrophication ultimately detracts from biodiversity, through the proliferation and dominance of nutrient-tolerant plants and algal species. These tend to displace more sensitive species of higher conservation value, changing the structure of ecological communities (Welch, 2004).

Eutrophication can adversely affect a wide variety of water uses such as water supply (e.g. algae clogging filters in treatment works), livestock watering, irrigation, fisheries, navigation, water sports, angling and nature conservation. It can give rise to undesirable aesthetic impacts in the form of increased turbidity, discolouration, unpleasant odours, slimes and foam formation (Tunney, 1997).

Problems of eutrophication are not only ecological. It has been estimated that in England and Wales eutrophication causes £75 – £114.3 million per year of damage to areas of freshwater. This is due to, for example, higher treatment costs for drinking water, reduced amenity value, reduced value of waterfront dwellings and loss of tourism. An estimated additional £54.8 million per year is spent in measures to address this damage (Pretty *et al.*, 2003).

2.0.1 Nutrient Sources

Non-point source pollution (alternatively known as diffuse pollution), occurs when there is no discrete point of discharge and pollution enters the environment by a multitude of pathways. Non-point source pollution can therefore be defined as follows:

"Pollution arising from land-use activities (both urban and rural) that are dispersed across a catchment, or sub-catchment, and do not arise as a process effluent, municipal sewage waste effluent, or an effluent discharge from farm buildings" (DEFRA, 2002b).

While the sources of non-point pollution may be man made, the environment itself often mediates their presence in the aquatic environment. Rainfall and the physico-geochemical properties of the land itself play a major role in determining the extent of diffuse pollution (Carpenter *et al.*, 1998).

To further protect water quality and to meet current and future legislation, diffuse sources of water pollution need to be identified, quantified and controlled. There are a wide variety of diffuse sources of pollutants to the aquatic system. These are generally dispersed and diverse in nature. Individually the sources may be small, but their collective impact can be damaging (Baker, 1992). Diffuse pollution can be derived from current and past land use in both agricultural and urban environments and can also include atmospheric deposition.

Diffuse water pollution is mainly related to the way land and soil are used and managed. Agriculture is a key generator of diffuse pollution, but it is not the only source and other land use activities also contribute to diffuse pollutant loads (Sagardoy, 1993). Activities such as forestry, industry, construction, urbanisation, transport and recreation may all contribute to the problem.

The main sources of phosphorus to surface water in the UK are presented in Figure 2.0.1, for 1993. It illustrates the significance of discrete point sources (41%), compared with diffuse sources (59%), of which 50% comes from agriculture and 9% is due to natural background levels (Morse *et al.*, *1993*).

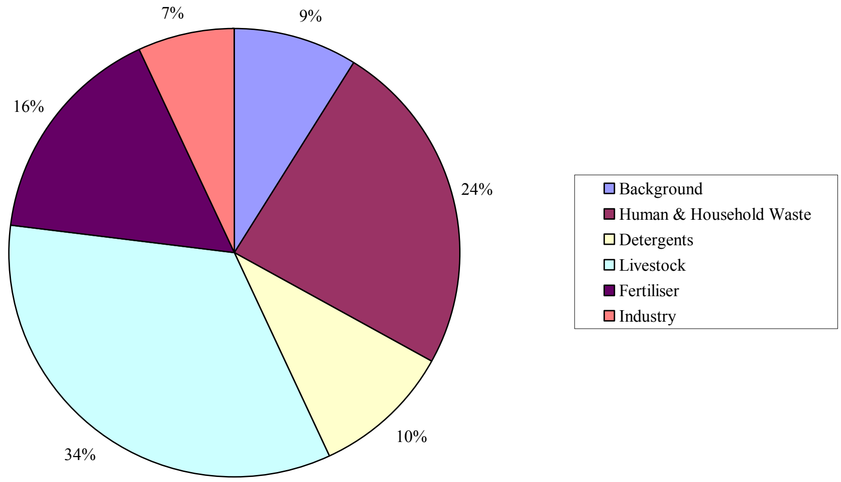
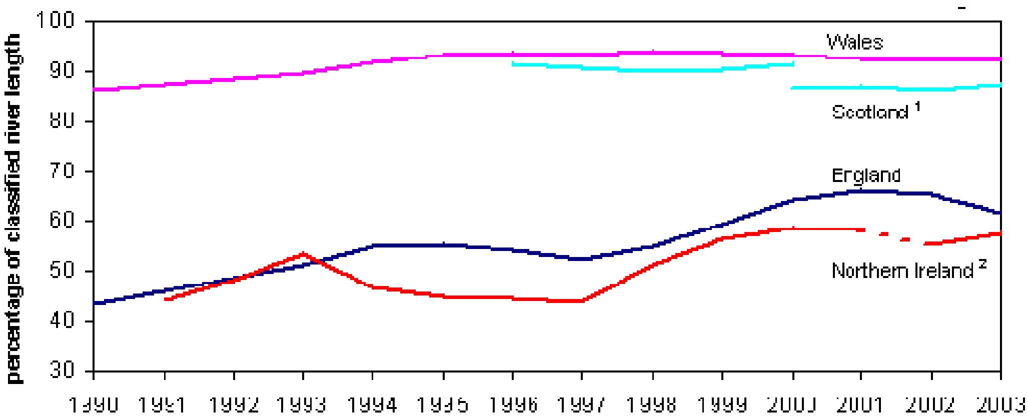


Figure 2.0.1: Major phosphorus inputs to surface waters in the UK (Morse *et al*., 1993)

Evidence suggests that diffuse pollution and its impacts on the wider environment are increasing throughout the UK. Johnes *et al*. (1994) estimated that total nitrogen and phosphorus concentrations in 10 English and Welsh rivers had increased by 62-372% and 53-171% respectively between 1931 and 1988. Similarly, in a selection of 94 UK lakes and reservoirs, it was estimated that total phosphorus loading had changed by between -89 and +9602%. Total nitrogen loads were estimated to have changed by between -78 and +9793% over the same period (1931 – 1988). Since these data were published, in terms of overall chemical water quality of rivers, there has been relatively little change in the proportion of rivers of good or fair quality in the UK since 2000, but there has been an improvement since 1990. Figure 2.0.2 shows that 62% of river lengths were of good chemical quality in 2003, compared with 43% in 1990 (DEFRA, 2004b).



1: Scottish river classification network changed in 2000

2: Northern Ireland classified network significantly expanded in 2002

Figure 2.0.2: Rivers of good chemical quality 1990-2003 (DEFRA, 2004b)

The amount and relative contributions of point and diffuse sources to total pollution loads varies between river basins, depending on the local biogeochemistry, geomorphology and the anthropogenic activities within (Bonta, 1998). For individual river basins these contributions will change with time. This can be both in the short-term, in response to land management and climatic variation, and in the longer term in response to technological advances, national and international regulations and voluntary codes of practice. Table 2.0.3 illustrates modelled nutrient budgets for four river basins in England, which show that agriculture contributes varying amounts of diffuse phosphorus in different river basins compared with point source phosphorus contributions from sewage treatment works (STW's).

Table 2.0.3: Annual Inputs of P (tonnes P year^{-1}) (Mainstone, 2000)

Source	**Upper reaches of Hampshire Avon**	**Warwickshire Avon**	**Pevensey Levels**	**River Ant**
Atmospheric/natural	12.5 (14%)	57.9 (5.5%)	0.6 (1.6%)	0.08 (1.6%)
Inorganic fertiliser	19.9 (22.4%)	209.5 (20%)	2.5 (7.4%)	1.04 (21.3%)
Livestock	18.7 (21%)	99.5 (9.5%)	2.4 (7%)	2.89 (59.3%)
STWs	35.5 (39.9%)	654.3 (62.6%)	28.7 (84%)	0.86 (17.6%)
Unsewered pop.		23.8 (2.3%)	-	-
Industry	2.5 (2.8%)	-	-	-
Total	**89.1**	**1045.0**	**34.2**	**4.87**
Catchment area (km^2)	1249	2892	56	49.3
P load exported to river (kg ha yr^{-1})	0.7	3.6	6.1	1.0

2.0.2 Nutrient Transfers

The role of hydrology in nutrient transfer is crucial as it provides both the energy and the carrier for nutrient transfer (Preedy *et al*., 1999). Without it there would be no transport mechanism from the hill slope to watercourse. Investigations of diffuse losses of nutrients from agricultural soils necessitate a clear understanding of the potential for nutrient transfer by surface and subsurface hydrological pathways.

Figure 2.0.3 illustrates the pathways from land to the aquatic environment, both via surface and groundwater. It illustrates the movement of pollutants from land to surface water by run-off and over land flow. Important processes here include erosion, leaching, adsorption and decomposition. Pollutants can also enter surface waters by interflow (subsurface flow) within the unsaturated zone in soils. In soils, pollutant concentrations are modified by adsorption, desorption, oxidation and decomposition. Once pollutants percolate to shallow groundwaters, interaction can occur between the groundwater and surface water systems. Finally, with percolation to deep groundwater, pollutants can cause long term contamination of water. It is important to note that pollutants generated on, or applied to, the land surface as a result of agricultural practices, move and/or undergo transformation reactions when water is present. Thus water is usually the main carrier or transporting medium of non-point pollutants (MAFF, 1998a).

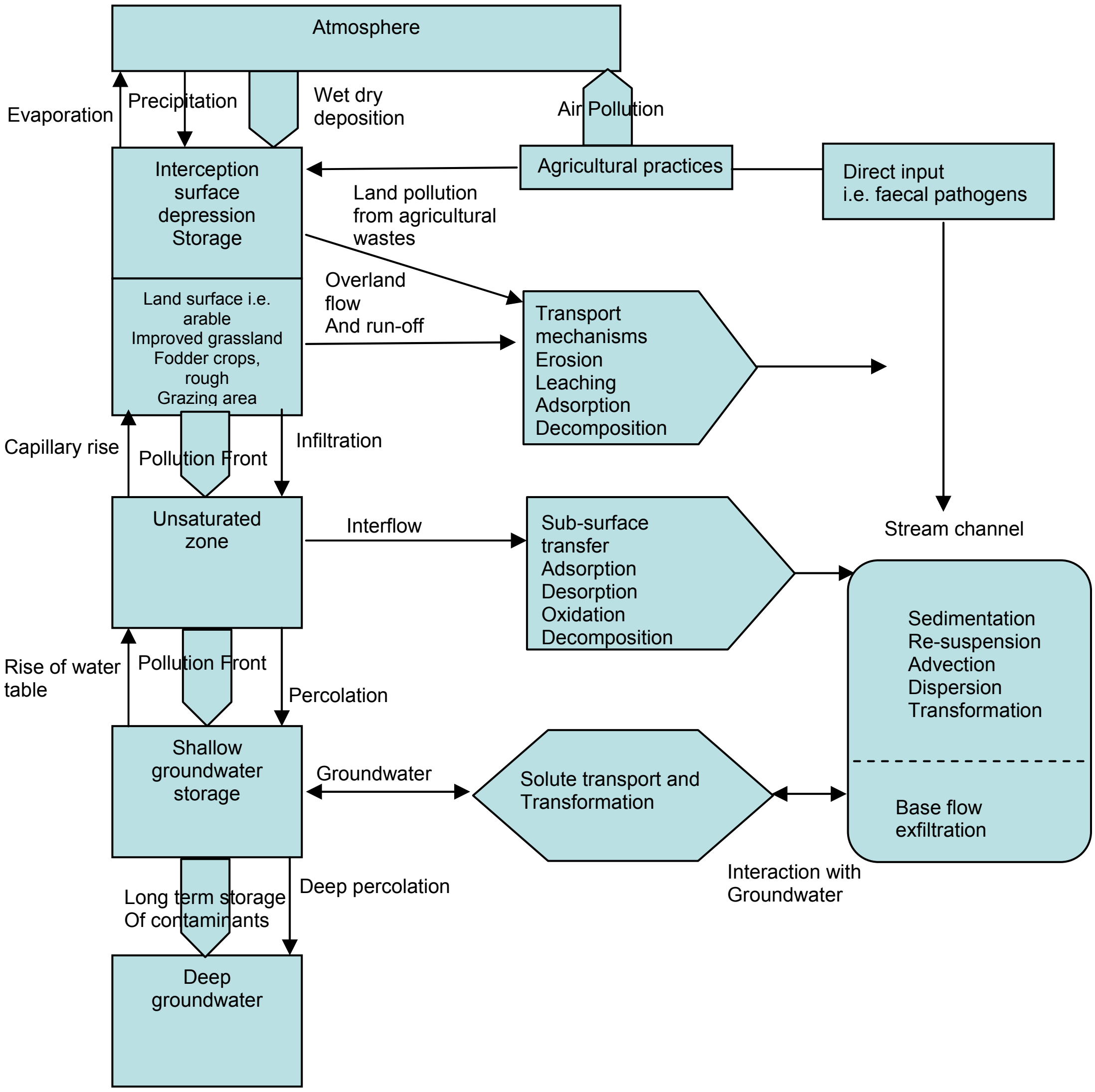


Figure 2.0.3: Diagram illustrating generalised losses of pollutants from land to water (Goulding, 2000)

Experiments in the UK have shown that most nitrogen is lost from arable farms by leaching during autumn and winter as nitrate (NO_3) and ammonium (NH_4) (mainly on sandy soils) (Hatch *et al.*, 2002). Denitrification is where NO_3 and nitrite (NO_2) are reduced to gaseous forms of nitrogen; nitrous oxide (N_2O), nitrogen (N_2) and nitric oxide (NO). There are two general hydraulic pathways by which mobile forms of nitrogen can be leached or transferred into waterways (Goulding & Webster, 1992). Horizontal flow occurs in soils with poor drainage. This may occur on the soil surface or above impermeable layers within the soil profile, such as the movement in duplex soil (sand over clay).

Losses of nitrogen by surface flow in eroded soils may contribute to nitrate loads, if the organic nitrogen present in the soil particles is released into watercourses then decomposes during or after the erosion process. This may happen many years after the erosion event. Sediments in lakes can release nutrients when disturbed by water movements in the lake as temperature changes and if the lake is dredged.

Vertical flow through the soil profile can be either via matrix flow through the whole soil body in tightly textured soils, or through bypass flow in large macropores and cracks in heavy textured soils. Retention times of mobile nitrogen within a soil are therefore dependent on many properties: for example texture, porosity and slope, or position in the landscape as well as the volume and frequency of rainfall events. For this reason the prediction of nitrogen leaching from soils is difficult (Stockdale, 1999).

The movement of phosphorus to surface and groundwaters occurs primarily in association with organic matter. Soil erosion by water and overland flow are widely recognised as the principal mechanisms by which phosphorus is removed from land (MAFF, 1998b). On poorly permeable soils or soils with poorly permeable sub-structure, mobilised phosphorus is likely to reach watercourses rapidly, either by surface run-off or throughflow in the soil, particularly via macropores or cracks.

The transport of phosphorus along throughflow pathways in agricultural river basins has been clearly demonstrated in a number of studies (e.g. Sharpley *et al*., 1981). In the UK, soluble reactive phosphorus release from adsorbed phosphorus in the soil is controlled by desorption kinetics. In soils where adsorption capacity has been reached or exceeded, additional inputs of soluble phosphorus from fertilisers may be directly leached through infiltration, together with the desorbed fraction, vertically down the soil profile away from the root zone. This load may reach subsurface drains through macropore flow, allowing rapid transport of soluble reactive phosphorus to adjacent watercourses along this subsurface quick flow pathway, bypassing soil adsorption capacity. Where soils are not under drained, soluble reactive phosphorus leached down to the soil profile where the lower organic content reduces adsorption capacity still further, can then follow a lateral pathway as through flow parallel to the hill slope, or vertically as percolation to groundwater. Therefore gradual phosphorus saturation of progressive horizons, leading to complete saturation of soil and groundwater adsorption capacity can be assumed, given long term application of phosphorus in excess of crop requirements and soil adsorption capacity.

The atmosphere can play a substantial role in the transport of non-point source pollutants. One of the first recorded examples of this was the international study of the Pollution From Land Use Activities Reference Group of the Joint Commission on non-point source pollution in the North American Great Lakes Basin (PLUARG (Pollution Land Use Activities Reference Group), 1978). A result of the PLUARG study was the finding that the majority of the annual phosphorus load to Lake Superior (the most pristine of the North American Great Lakes) entered via atmospheric deposition. Elsewhere, Balon and Coche (1974) have noted a similar role for the atmosphere relative to nitrate-nitrogen inputs to Lake Kariba (Zimbabwe/Zambia).

The importance of flux of pollutants from the atmosphere directly onto the surface of land and inland water depends primarily on the magnitude of the pollutant load from other pollutant sources in the drainage basin. Wet deposition is a function of contaminants carried to water surface by precipitation. Dry deposition is the common name for all deposition processes other than wet deposition. It includes transfer of gases from the atmosphere to vegetation, soil, lake water etc. and may involve both biotic and abiotic fixation of the gases. Large particles will fall by gravity; atmospheric movements must transfer smaller ones to the surface where they can be captured by impact, interception or molecular diffusion through the layer nearest the surface. As with gases the deposition rate depends on the surface topography and atmospheric turbulence but also particle size. Dry deposition rates are dependent on particle size distribution.

2.0.3 The Impact of Land Use on Nutrient Supply and Transfer

All forms of land use potentially can affect the quality of run-off or leaching from the land surface to receiving waters. In an undeveloped area naturally occurring physical, chemical and biological processes interact to recycle most waterborne materials in run-off (Gburek & Sharpley, 1998). When a river basin becomes more developed these processes, which ameliorate the potential environmental effects of natural pollutants, are disrupted. Humans contribute to this disruption by adding polluting materials such as fertilisers to the land surface. Run-off will wash these materials off the land surface, or when leaching moves them through the soil, the non-point source pollutant loads carried to receiving waters can increase significantly (Kwaad, 1991).

The term ‘land use’ refers to the purpose for which a given area of land is being used (Ryding & Thornton, 1999). The concept of land use is fundamental to assessing non-point source pollution, primarily because it denotes the activity on land (or use of land) that generates pollutants of concern. In its simplest designation, land can be categorized as forest, agriculture or urban. There are many delineations and sub-delineations of land usage, which may need to be considered along with the atmosphere as a transport pathway and a source of non-point nutrient pollution. Table 2.0.4 provides the general characteristics of nutrient generation from specific land uses.

Although the atmosphere has been listed as a source of nutrient contamination, in fact, with the possible exception of some specific chemical reactions related to the hydrolysis of acidic emissions from industrial sources, the atmosphere does not constitute a ‘source’ of any pollutants (Ryding & Thornton, 1999). It should be viewed instead as a transport pathway for pollutants released to the atmosphere.

Table 2.0.4: General characteristics of nutrient generation from specific land uses (Reckhow *et al.* 1980)

Land Cover	Factors
Forest	• Forest river basins with sandy soils overlying granitic igneous river basins produce half the phosphorus export of forest river basins with loam soils overlying sedimentary formations • Young (< 5 years) forests produce greater water and sediment-phosphorus run-off than old forests • Areas with warm climates and high rainfall produce greater water and higher phosphorus export than other climatic regions • Deforestation or timber harvest activities generally cause greater nutrient export than undisturbed forested river basins • Forest fires cause a temporary increase in phosphorus export; however, the levels usually return to pre-fire conditions relatively quickly
Agriculture	**Cropland** • Phosphorus export via run-off from sandy or gravel soils is generally small • Phosphorous export in run-off from clay soils and organic soils generally is large • The time of fertiliser application is critical; for manure-type fertilisers applied to frozen soils, the phosphorus export following snowmelt and high rainfall periods is large; incorporating applied fertilisers into the soil results in reduced nutrient export • Fertiliser application above recommended levels for existing soil conditions produced increased nutrient export • Conventional tillage methods (e.g. fallow land during non-growth seasons, harvest removal of crop residues) results in large nutrient export • Conservation tillage practices reduce nutrient export; however, they can result in increased herbicide usage and export • Row crops produce much larger nutrient exports than non-row crops
	Pasture • Limiting livestock grazing time on a given parcel of land reduces nutrient export, in contrast to continuous grazing on the same parcel • Fertilised pastures often exhibit increased nutrient export • The greater the animal density, the greater the potential for increased nutrient export
	Feedlot and manure storage • The greater the extent of impervious surface, the greater the nutrient export • The greater the animal density, the greater the nutrient export • The greater the roof area: feedlot area ratio, the smaller the nutrient run-off • The use of a detention basin decreases nutrient export
Urban	• The greater the extent of impervious surface, generally the greater the nutrient export • Increases in housing density, fertiliser applications, and pet density result in increased nutrient export • Decreases in grass and vegetative cover result in increased nutrient export • Commercial, business, and industrial areas often have considerable vehicle/ pedestrian traffic, resulting in greater nutrient export than residential areas
Atmosphere	• In agricultural areas, increased nutrient export can result from ammonia volatilization from feedlots and fertilisers, and from wind erosion of fertilised soils • Increased nutrient loads from agricultural areas often coincide with fertilisation and tilling periods • Increased nutrient export can result from boiler and furnace operations, and from automotive and aviation emissions, in urban areas

2.0.4 The Role of Climate Change

The natural background stream water chemistry is determined by atmospheric inputs, river basin geology and soil type, the process by which water reaches the river network and by chemical and biological processes operating in the river. Superimposed on these natural controls are anthropogenic factors such as river basin land use, deposition of atmospheric pollutants, the discharge of used water from sewage treatment works and other uses, drainage from urban areas and pollution incidents. The effects of climate on natural or chemical processes will be superimposed on these anthropogenic factors, which are also varying through time for many reasons, including climate.

Temperature is perhaps the most important physical quality characteristic of river water. It not only affects biological and chemical processes in the river, but also influences aquatic ecosystems. The rates of biological and chemical processes are temperature dependent. Temperature influences the ability of water to absorb gases such as nitrogen, oxygen and carbon dioxide.

Increases in temperature can increase the rates of both nitrification and denitrifaction, but denitrification rates are most affected so, other things being equal, high water temperatures would lead to a reduction in nitrate concentrations (Jenkins *et al.*, 1993). Increased residence times, due to lower flows, would also result in lower nitrate concentrations because denitrification could continue longer. These effects might be offset by the reduced dilution due to lower flow volumes: the actual change in nitrate concentrations in a river will therefore depend on temperature change, the change in stream flow volumes and the volume of nitrate inputs.

Jenkins *et al.* (1993) simulated nitrate concentrations and temperature change in a number of rivers and showed that nitrate concentrations are reduced, particularly during the summer. There are three other important potential effects of increased temperatures. First, a change in agricultural practices triggered by climate might lead to a change in inputs of agro-chemicals. Second, higher temperatures and drier soils would increase the rate of mineralization of organic nitrogen in the soil, increasing the amount of nitrogen available to be washed into the rivers. Third, peak nitrate concentrations occur in autumn when nitrates that have accumulated in the soil through prolonged summer dry spells are flushed into the river. Taken together these three influences might lead to an increase in nitrate concentrations and more particularly to an increase in peak concentrations.

Increased temperatures are likely to increase the duration of stratification in lakes and by changing rates of bacterial activity will alter dissolved oxygen (DO) levels. For example, Blumberg and DiToro (1990) simulated a reduction in DO in Lake Erie of 1 mg l^{-1} in the upper layers and up to 2 mg l^{-1} in the lower layers for a rise between 3.5 and 4.3 degrees in air temperature. Stefan and Fang (1994) found larger changes in a number of smaller lakes in Minnesota and simulated reductions in temperature at depth in some lake types. In many lakes DO levels at the surface are close to saturation and so would change relatively little as temperature rises. Shallow lakes do not develop thermal stratification. Given an increase in summer radiation, water temperatures in such lakes are likely to rise and DO concentrations will fall.

Changes in catchment vegetation, rainfall and hydrological regimes will affect erosion on hill slopes and in river channels, sediment transport and deposition, and river channel stability. There have been very few studies into the implications of global warming and it may be difficult to separate climate change effects from the consequences of land use change.

Boardman *et al.* (1990) simulated the effects of changes in rainfall regimes on soil erosion in Britain, assuming no change in land use. Increased winter rainfall resulted in greater erosion from arable fields in lowland Britain, but lower losses in upland Britain because warmer temperatures meant that ground cover lasted longer through the winter. Not all the sediment created by soil erosion reaches the river network, so the implications for stream sediment yields are difficult to estimate.

The effects of global warming on stream and lake water quality are much less well understood than effects on water quantity. It is clear that changes will depend on local climatic, geological and hydrological conditions as well as on the environmental control measures in place. There appears to be a general deterioration in water quality with higher water temperatures, especially in lakes and rivers, which already receive high effluent inputs.

2.1 Eutrophic Lakes

Although it has been recognised that eutrophication affects all water bodies the main focus of this thesis is that of lakes. All further discussion will be restricted to lakes. Eutrophication research has had a high profile since the late 1980's. The widespread occurrence of blue-green algal blooms in standing slow-flowing fresh waters gave rise to considerable interest and concern by the public, the media and within the water industry (FWR, 2000). The relation between nutrients and lake eutrophication is reasonably well understood. In the past 5 years it has been recognised that many more factors such as light, temperature, flow regime, turbidity, zooplankton grazing and toxic substances are affecting the eutrophication process in addition to, or maybe instead of, the nutrients (Van der Molen & Boers, 1999). However, not much progress has yet been made in the quantification of these factors.

Eutrophic lakes are typically small, shallow and rapidly flushed. It has been estimated that in the northeastern states of America 65% of lakes, which are eutrophic, are less than 23 hectares in size, shallower and in river basins with more human activity than natural lakes (US Environmental Protection Agency, 2002). A similar situation can be seen in Italy, where most of the lowland lakes subject to more or less severe eutrophication processes occur in highly populated and industrialised areas. This is attributed to high point source loading in the past and high nutrient input from arable land at present (Institute of Ecosystem Study, 2002).

In Denmark the majority of lakes are highly eutrophic due to high nutrient input from domestic sources and agricultural activities. Here, attempts have recently been made to reduce nutrient loading on lakes by intervening at the source level and by improving the retention capacity of river basin areas (Jeppesen *et al.*, 1999). External loading of some Danish lakes is now so low that a shift to clear water state ought to have taken place. Slow recovery is sometimes observed due to both chemical and biological factors. Phosphorus release (internal loading) from the phosphorus pool accumulated in lake sediments during the time when loading was high may counteract the reduction of external loading. A good example is the shallow Lake Sobygaard in which internal loading was still high 13 years after a 90% reduction in external loading (Sondergaard *et al.*, 1993).

Internal loading problems can also be seen in Lake Muggelsee in Germany. Lake Muggelsee was highly loaded with phosphorus and nitrogen by the River Spree up to the end of the 1980's. At the end of the 1980's, the phosphorus retention capacity of the sediment was exceeded and the lake became a source of phosphorus. The lake regained its ability to retain phosphorus in the sediment after external load reduction in the 1990's. However, the internal load of phosphorus still remained high, off setting any change in phosphorus load expected by external load reduction (Kozerski *et al.*, 1999).

Both external and internal loading problems have occurred in the Naardermeer Nature Reserve in the Netherlands, where during the last century the chemical composition of the surface water has changed from brackish to fresh and from unpolluted to nutrient rich. Lake Grote Meer contained <0.03 mg l^{-1} PO_4, < 0.1 mg l^{-1} NO_3 and 0.1 mg l^{-1} NH_4 in 1908 (Bootsma *et al.*, 1999), which in 1985 had increased to 0.05, 0.38 and 0.18 respectively as a result of both internal and external inputs (Barendregt *et al.*, 1995). Internal sources were guano from the Cormorant colony nearby, mineralization of the peat soil, which was due to the lowering of the water table and nitrogen fixation by alder. The main external nutrient sources were from local sewage treatment works and agriculture.

In the UK similar problems with eutrophication have been occurring. Currently there are 86 rivers and canals, 10 estuaries, 16 lakes and reservoirs, which have been identified and designated Sensitive Areas (Eutrophic) under European legislation (Newson, 1991). One such site is the Broads.

2.1.1 Eutrophication in the Broads

The Broads were created in medieval times by peat digging (Lambert & Jennings, 1960); man from then to the present day has influenced the area. Until the turn of the 20th century the rivers were important arteries for local transport, but with the development of railways their value as a recreational resource began to be recognised. Many naturalists have provided documentary evidence of the wealth and diversity of the biota found in the Broads (George, 1992).

By the late 1960's surveys revealed that the now highly eutrophic Broads had changed considerably (Mason & Bryant 1975) from the state described at the turn of the 20th century. The once clear water, dominated by a variety of submerged aquatic plants, including a diverse array of charophytes, had almost completely disappeared. Associated with this loss was a dramatic reduction in invertebrate diversity and an improvised fish community of low biomass dominated by small roach (*Rutilus rutilus*) and bream (*Abramis abramis*) (Phillips *et al.*, 1999). There is also evidence of an increase in salinity of the water some years prior to and perhaps during these changes because of more powerful land drainage pumps which were installed in 1939 (Watson, 1981) to the immediate river basin and potentially exploited a deeper more saline water table.

The coincidence of these changes in aquatic ecology and water quality suggest that they are inter-related and their magnitude indicates a potent cause rather than some subtle progressive natural change (Moss and Leah, 1982). Two likely causes have been identified: eutrophication and salinity changes. The system has been brackish for many decades, probably for two centuries, and it is unlikely that chloride changes in an already brackish system would have caused the major loss of aquatic plants. Lakes elsewhere also support rich charophyte and other aquatic plant communities at chloride concentrations far higher than those seen in the Broads e.g. the upper Baltic Sea and Loch Obisary (Moss & Leah, 1982). The change in plant dominance is more likely to have resulted from eutrophication rather than salinity changes.

Eutrophication of the Broads can be described in three phases, each relating to different nutrient levels. These phases are conveniently defined by the level of phosphate in the water, since phosphate is a critical and often the limiting nutrient for algal and plant growth (Broads Authority, 1982).

Phase 1 (Pristine State): This is the situation that existed in most of the Broads prior to 1900. In this condition the Broads resembled the 'marl lakes' which occur on chalk, limestone or on fluvial deposits derived from these rocks. An example of this is the Malham Tarn in North Yorkshire and the Durness Lochs in the Highland region of Scotland (George 1992).

The vegetation was fairly sparse and of low growing species. Stoneworts and the Bladderwort, which favour relatively infertile water, but which, unlike algae can take phosphorus they require from sediment, would have been prominent. Fossil diatoms from cores suggest that epiphytic and bottom living algae were scarce and phytoplankton virtually absent (Moss *et al*., 1996). The water therefore would have been crystal clear.

Studies have shown that prior to 1800 Barton Broad had a total phosphorus loading of about 0.4g m^{-2} y^{-1}, and that the mean concentration of total phosphorus was therefore only 13.3 µg P l^{-1}. A level of fertility of approximately 3% of the mean of about 360 µg P l^{-1} recorded at this site in the 1970's (Moss, 1980). Data obtained from cores taken from Belaugh, Hoveton Great and Cockshoot Broads indicate that during the nineteenth century, the mean total phosphorus levels in these sites and the River Bure, with which they are connected, were similar at this time to those of Barton Broad. Unfortunately none of the Broads today still possess a true Phase 1 flora.

Phase 2: The second phase was marked by a gradual increase in the nutrient loadings of the rivers and Broads. This was due to the increasing efficiency of farming methods and the increasing quantities of raw sewage being discharged into the rivers. The higher nutrient levels encouraged a more luxuriant growth of taller plant species. Those successful in the Broads include Horned Pondweed, Water Soldier and Yellow Water Lily, particularly the Spiked and Whorled Water-milfoils and Fennel-leaved Pondweed. The robustness of these nutrient demanding species gave them a competitive advantage over the waterweeds of Phase 1.

Sedimentation cores show that the mean total phosphorus loading of Barton Broad increased from 1.55 g m^{-2} yr^{-1} in 1900 to 2.15 g m^{-2} yr^{-1} in 1920. From these figures it can be calculated that the mean total phosphorus concentrations in the water increased from 52 – 72 μg P l^{-1} (Moss, 1980). Evidence suggests that the transition from Phase 1 to Phase 2 in the River Ant and the River Bure occurred between 1850 and 1890. In the Thurne Broads the Phase 1 dominated flora persisted longer, with the transition taking place in the mid 1930's. Ten different species of Chara were still recorded in these Broads at the turn of the 20th Century. Six of these species were still present in 1960 but by then were growing in association with species typical of Phase 2 flora (Jackson, 1978).

Phase 3: Most of the open waters in the region are now in this phase. The high nutrient level of the water resulted in a substantial loss of diversity in the benthic invertebrate fauna. An accelerated rate of sediment deposition could also be seen along with a large increase in the amount of phytoplankton in the water and eventual disappearance of the waterweed flora.

The switch between Phase 2 and 3 was assumed to be a relatively simple process and was attributed to the increased phosphorus loading of the rivers. It has however been found that the transition from Phase 2 to Phase 3 is complex and subtle (George, 1992). It is thought that Phases 2 and 3 are alternative, relatively stable communities at high nutrient concentrations with self regulatory mechanisms existing which buffer the effects of extreme change (Moss *et al*., 1996). One of these buffers is the heavy predation to which cladocerans are subject in the absence of water weeds. Cladocerans graze on algae, with increased nutrient loads algal populations will increase. If cladocerans are subject to heavy predation from fish then algal populations cannot be controlled resulting in turbid waters (Moss, 2001). This can result in the consequent tendency for Phase 3 community to remain in this state once the switch from Phase 2 has occurred.

Currently 53% of the Broads are in fair (Phase 2) ecological condition (Broads Authority, 2003). These include Hickling, Horsey and Martham South Broad. All three rivers in the study area (Bure, Ant and Thurne) are designated as sensitive areas (eutrophic) under the Urban Waste Water Treatment Directive (Moss, 2001).

Nutrient inputs to land from fertilisers and animal wastes in the Anglian region are amongst the highest in the UK (Johnes, 1997; Johnes *et al*., 1996a). Under the Water

Resources Act 1989 all sewage treatment works and industrial sources receive phosphorus removal treatment before being discharged into the River Ant. All the major sewage treatment works discharging into the River Bure also receive phosphorus removal treatment before entering the water course (Moss, 2001). There are still many sewage treatment works that do not fall under the Water Resources Act, due to their size. When these two nutrient inputs are combined, sewage treatment works and septic tank facilities are combined with nutrient inputs from land. These comprise a large potential source of nutrient export to adjacent rivers and lakes. The nutrient enrichment of the major rivers feeding into the Broadland ESA has been studied for some time (Moss *et al*., 1984; Moss *et al*., 1988). The impacts of these increases in nutrient loading on the Broads are well known (Moss *et al*., 1985).

There is a perceived need to restore their ecological status, the Broads have become a classic site for research on bio-manipulation as a technique for the restoration of the ecological status of eutrophic standing waters (Phillips *et al*., 1994). This technique holds great potential as a restoration tool, but is unlikely to achieve a sustainable plant dominated, clear water state in the Broads unless nutrient loading on the system can be reduced in tandem (Johnes, 1996b).

2.2 Eutrophication Management

Eutrophication has been recognised as a problem throughout the world for many years (Banens & Davis, 1998). A huge number of fresh and marine water areas, of varying sizes, have been seriously affected by eutrophication. This is a result of increasing discharges of phosphorus and nitrogen from modern society (Vollenweider, 1981). The large-scale decline of many aquatic ecosystems due to eutrophication has been recent and precipitous. In the Black Sea, for instance, severe eutrophication has a history of about three decades. This has been attributed to increased standards of living and food production which led to a high increase in nutrient discharge to rivers with subsequent algal blooms (Mee, 1997). In Lake Victoria, the world’s second largest freshwater lake, eutrophication is a result of increased nutrient levels since the 1950’s, when population started to increase along the shores of the lake and in its river basin area (Nyirabu, 1997).

The development of large scale eutrophication in other parts of the world is in principle regulated by the same factors as those causing deterioration in Lake Victoria and the Black Sea, namely:

- A growing population and urbanization of the river basin areas
- Increased standards of living and food production including several fold growth in use of fertilisers and irrigation in agriculture (Forsberg, 1998).

In order to control eutrophication it is of central importance to know which nutrient is most limiting for algal growth. Comprehensive limnological research during the 1960's and 1970's demonstrated with nutrient enrichment experiments, nutrient supply ratios, and relationships between nutrients and algal biomass, that phosphorus is most often the limiting nutrient in fresh water (Vollenweider & Kerekes, 1982a). In the marine environment e.g. the Baltic Sea, nitrogen is regarded to be more important than phosphorus (Graneli *et al.*, 1990). But during the summer, blooms of cyanophytes have the ability to fix atmospheric nitrogen and therefore phosphorus may also act as the limiting nutrient here. In the Broads, there is still debate on which nutrient is the most important for limitation of algal growth.

In Denmark the debate over nitrogen versus phosphorus as the limiting factor of algal growth caused the Danish Government to make a far-reaching decision. In 1987 the Danish Government decided to limit the pressure on the aquatic environment by reducing the loads of phosphorus and nitrogen by 80% and 50% respectively (Harremoes, 1998). In Denmark the reduction of the phosphorus load has been successfully accomplished by government legislation and the reduction of nitrogen from wastewater has been achieved; but the reduction of diffuse sources has not been very successful.

The lake restoration strategy in the Netherlands has also focused on the reduction of external nutrient loading. Standards for general environmental quality were set at 0.15 mg l^{-1} for total phosphorus and 2.2 mg l^{-1} for total nitrogen. So far the control of external loading has not resulted in the water quality desired (Hosper, 1998).

Turbid lake ecosystems tend to be resistant to recovery and solely reducing the external nutrient loading seems to be insufficient for attaining clear water conditions. It has been found that the reduction in external loading may be counteracted by internal loading, giving a marginal response in water quality (Shapiro, 1980). It is necessary to enforce a shift from the turbid to the clear state using quite rigorous methods.

Biomanipulation, the removal of a large part of the fish population from the system, has been applied in various lakes. By removing large fish from a turbid lake, effects may ‘cascade’ through the trophic system, resulting in a switch to the clear water state. Unfortunately almost half of the biomanipulations in the Netherlands were unsuccessful due to too high external nutrient levels (Coops, 2004).

Phosphorus treatment was attempted in the Eau Galle Reservoir (Wisconsin, USA) in 1986. Internal loading was three to six times greater than external loading (James *et al*., 2003). This was undertaken by hypolimnetic injection of alum. Internal loading dropped substantially in the first year after treatment. Exceptionally high external loads increased phosphorus levels, thus negating the reduced internal loading by the second year. High external loading of nutrients hampered the successfulness of eutrophication management in the Eau Galle Reservoir, as also highlighted in the Netherlands

Work carried out by Edwin *et al*., (1998) in the Netherlands suggests more specific standards should be set that are water type dependent. Table 2.2.1 shows standards proposed by Edwin *et al*., (1998) these standards have yet to be incorporated into any Government policy and do not include standards for lakes. The standards were defined as a list of physical and chemical values and by a verbal description of the biotic community that had to be present.

Table 2.2.1: Proposed Standards for Nutrients (Edwin *et al.*, 1998)

Watercourse description	Orthophosphate (mg P l^{-1})	Total phosphate (mg P l^{-1})	Nitrite – nitrate (mg N l^{-1})	Ammonium (mg N l^{-1})
Hill stream upper reach	0.08	0.24	4.95	0.20
Hill stream middle reach	0.54	0.72	4.24	0.30
Hill stream lower reach	0.80	1.00	4.65	1.30
Lowland stream upper reach	0.06	0.15	2.40	0.14
Lowland stream middle reach	0.14	0.18	5.64	0.37
Lowland stream lower reach	0.19	0.36	5.00	0.70
Sandy bottom ditch	0.05	0.08	0.34	0.27
Clayish bottom ditch	0.08	0.17	0.89	0.16
Peaty bottom ditch	0.05	0.14	0.11	0.20
Acid ditch	0.05	0.05	0.14	0.05
Brackish ditch	0.20	0.42	2.30	1.80
Slightly brackish ditch	0.82	1.90	1.28	3.40

It can be seen that the restoration of lakes requires two phases, each addressing a different source of nutrients in the lake. The first phase needs to be river basin wide to deal with nutrient loading from external sources. The setting of a single water quality standard by Governments does not guarantee the minimum quality level in watercourses to protect the relevant aquatic communities.

The second phase cannot be implemented until the first phase has been completed. In the second phase, methods are employed to reduce the recycling of nutrients from sediment into the water column, such as biomanipulation, dredging, lake drawdown, aeration and chemical treatments (Beegle *et al.,* 2000). River basin wide issues that affect water quality must be addressed and external loading reduced before in-lake treatments are done.

A restoration programme for a specific lake starts with a definition of the targets or objective for the lake e.g. a natural reference state may often be included in the objective for lake restoration. Even if a natural reference state is known and maybe even achievable, another reference state may be preferred. Many birds inhabit for example man-made polders in the Danube Delta, Romania. Taken with the remaining reed-areas, these have increased the number of habitats available (Molen & Boers, 1999). The original natural state is not always preferred even from a natural reference point of view.

Several factors affect the feasibility of a natural reference and derived water quality targets: the present and future functions of the lake are often related to this along with hydrological and nutrient constraints. Functions like recreation and commercial fisheries limit the ecological possibilities of a lake. Disturbance is detrimental to several bird species and mammals (Benndort, 1987). Recreation and other water transport can seriously affect the shoreline of water bodies and their habitats and will have an impact on the re-suspension of sediments. It is clear that most functions of a water body other than ‘nature’ conflict with ecological water quality targets.

The effects on lake management are related to the condition of the system. Based on the costs of restoration, protection of lakes not affected by pollution should have the highest priority. Re-establishing the ability for self-restoration can combat minor influences on lakes. When the conditions can be created for a natural recovery, patience is preferred over a hasty approach with additional measures to achieve the target. For most lakes in the Netherlands lake re-creation is necessary. Due to many factors affecting the lake water quality, water managers should be aware that the re-creation of a more beneficial system is a more realistic approach than to re-create the pristine state (Schot *et al.*, 1998).

2.2.1 River basin Management

Water resource and land use planning can no longer be undertaken in isolation. This has been highlighted in the recent EU Water Framework Directive (WFD), which aims to understand the implication of integrating policies with regard to land use and diffuse pollution and point source regulation (Cleverly, 2001). River basin management involves the linking of physico-chemical attributes of water with ecological targets and the requirement for cost: benefit analysis as a key component of sustainable management. Other forms of legislation and restrictions on land use, such as the Nitrates Directive and protected areas also impact on river basin management (Environment Agency, 2003).

The river basin provides a unique spatial context for linking processes within and between ecosystems (upland, lowland and coastal). In particular, the hydrological cycle links together the atmosphere, biosphere and geosphere and water quality and water resources are fundamentally influenced by biogeochemical reactions within soils and the landscape (Roberts & Coutts, 1997). The underpinning geochemical reactions and sink-source relationships for nutrients and pollutants can be heavily influenced by management and manipulation that can influence the quality and ecology of both surface and ground waters (Queensland Department of Primary Industries, 1991). River basin management requires an understanding of the complex interactions between water and habitat quality and the various impacts arising from the often conflicting needs and pressures from water and land users (Fig 2.2.1).

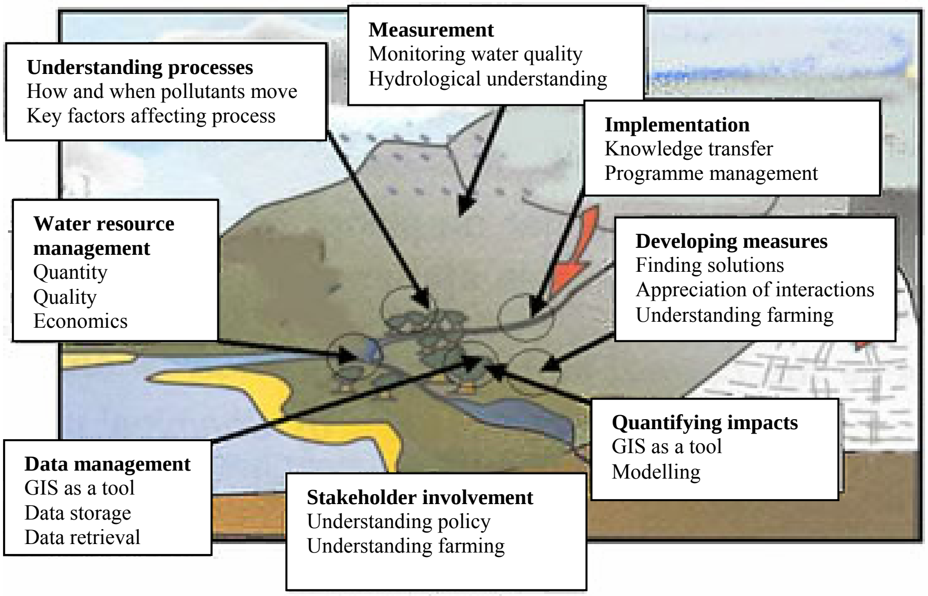


Figure 2.2.1: River basin Management (ADAS, 2004)

There are many different ways of approaching river basin management. In central Europe the most common technique is landscape planning with very exact rules that are fixed in legislation (see Bastian & Schreiber, 1999). In the USA, river basin management is more flexible and based on the analysis of actual conditions (Peterjohn & Correll, 1984).

In Australia river basin management issues are addressed through a community wide program called LANDCARE. This programme has strong government support at all levels and promotes more sustainable land and water management. This is achieved through a balance between economics, ecology, productivity and resource protection whilst contributing strongly to community development. LANDCARE has over 4500 autonomous community groups. Most are comprised of farmers and other landowners restoring land and increasing sustainability (Youl *et al.*, 2003).

LANDCARE partnership projects tackle salinity, revegetation, wildlife rescue, vegetation management, environmentally sustainable development, research and wetlands. One such project is the Riverglades Wetland Complex in South Australia. The Riverglades are being transformed from a degraded area into a functioning and healthy wetland with improved water quality and biodiversity. The project involves a feasibility study into infrequent drying cycles to control carp and plague minnows and to trigger regeneration of native plant species (Zukowski, 2001).

In New Zealand a lot of research has gone into the management of the impacts of both natural and anthropogenic change on land and water resources. Early on the value of whole-river basin studies as a means to quantify these impacts was recognised. Integrated Catchment Management (ICM) is a widely used tool throughout New Zealand. This is an approach which recognises the river basin as the appropriate organising unit for research on ecosystem processes for the purpose of managing natural resources in a context that includes social, economic and political considerations (Bowden, 2000). One such project is in the Motueka and Riwaka river basins; the goal of this program is to use historical research, biophysical experimentation, simulation modelling, and social learning to address resource management issues. Like LANDCARE projects in Australia this project relies on community groups, landowners and farmers due to its multiple, interacting and potentially conflicting land uses (Basher, 2003).

A relatively recent technique is assessment of critical source areas (Behrendt *et al*., 1996; Sharpley *et al*., 1994 & Pionke *et al*., 2000) to determine areas in the river basin where possible nutrient leaching and run-off are higher. This approach is based on the detailed spatial analysis of the river basin, where all natural and human impact conditions are considered, to determine sources of nutrients. Main factors affecting the nutrient transport are land use and soil cover (Sharpley *et al*. 1994; Pionke *et al*. 2000). Spatial analyses allow the identification of areas where nutrient losses are expected to be higher and where river basin management measures should consequently have more effect.

To decrease nutrient losses from river basins different mitigation measures can be used. The most common measures that are widely used in many countries, are a design of landscape structure with proper relation of natural and cultivated areas with connecting corridors (Forman & Gordon 1986; Haycock & Muscutt 1995; Collinge 1996), buffer zones (Lowrance *et al*. 1984; Peterjohn & Correll 1984; Pinay & Decamps 1988; Mander *et al*. 1995), constructed wetlands (Fleischer & Stibe 1991; Arheimer & Wittgren 1994; Straškraba 1996; Blackwell *et al*.1999; Kuusemets &

Mander 1999), and river restoration (Petersen *et al*. 1992). Storage lakes are also considered as possible nutrient sinks (Straškraba 1996; Jensen & Skop 1998; Mander & Järvet 1998). These storage lakes can be filled with sediments and saturate with nutrients and become sources of secondary pollution if they are not cleaned regularly.

2.2.2 Agricultural Management

A second method for addressing diffuse pollution is to reduce the transportation of pollutants from the field to watercourses by changing agricultural management. The activities that increase the availability of the pollutant, or cause soil release, are mechanical agitation of the soil (such as ploughing), removal of ground-cover plants and applications that are badly timed relative to weather or nutrient demand by the crop (Kuusemets & Mander, 2002). Activities that increase the rate of run-off are soil compaction caused by machinery, furrows (tramlines) that channel water flow towards water bodies, drainage, over irrigation and depletion of organic matter in the soil (Santhi *et al*., 2002).

There have been steady gains in the adoption of conservation tillage by farmers. In 1983, 23% of all the cropland in the USA was under some form of conservation tillage and in 1993 the percentage increased to 37% (Bull and Sandretto, 1996). There are many different forms of conservation tillage. Examples include no tillage, mulch tillage and other tillage operations that leave crop residue on the soil surface. The main benefit of conservation tillage is the protection provided to the soil by the crop residue. The crop residue reduces the detachment of soil particles by rainfall impact. Conservation tillage is classified as a source reduction and managerial practice that reduces sheet and rill erosion (Mostaghimi *et al.*, 1992). Researchers have reported reductions of up to 50% with 9-16% increase in crop residue coverage (Baker and Laflen, 1982). Other benefits of conservation tillage include: increased infiltration (Doa, 1993), protection from wind erosion (Blevins and Frye, 1993), reduction in evaporation (NRCS, 1998), increased soil organic matter and improved tilth (Hubbard and Jordon, 1996).

Although conservation tillage is very effective in reducing erosion, there are some concerns that it may increase potential pollution by other transport processes.

Conservation tillage increases infiltration and the potential for leaching of dissolved chemicals (Drury, 1993). Under conventional tillage, fertiliser or manure is incorporated into the soil by direct injection or by tillage operations. Both of these operations incorporate the crop residue. Under conservation tillage the manure or fertiliser is usually applied to the soil surface and not incorporated to minimise residue disruption. The nutrients tend to accumulate near the soil surface (Erbach, 1982). The increased nutrient level at the soil surface leads to increased nutrient concentrations in surface run-off (Baker and Johnson, 1983).

A further effective erosion control practice on low to moderate sloping land is contour farming. Contour farming is defined as farming sloping land in such a way that land preparation, planting and cultivating is done on the contours (NRCS, 1998). Contour farming provides protection against sheet and rill erosion. The greatest protection is provided against storms of moderate to low intensity on fields with mild slopes. A shortcoming of contour farming is that it provides minimum protection against high intensity storms on steep slopes. When storm intensity greatly exceeds the infiltration rate the accumulation of water behind the furrows many lead to ‘overtopping’ (Medez-Delgado, 1996). Overtopping occurs when ponded water overtops the furrow and from one furrow to the next creating a cascade of failures. This failure may result in severe local erosion in the form of gullies. Overtopping can also occur for storms of moderate intensity if contour farming is used on steep fields (Heatwole *et al.*, 1991).

Contour farming is generally used as a component of other practices, such as strip cropping and terraces. Strip cropping on the contour allows for the application of contour farming on steeper slopes. The closely spaced crops used in strip cropping reduce the potential for overtopping.

Buffer zones or filter strips reduce the transport of pollutants and are considered structural practices. They are planted or indigenous bands of vegetation that are situated between pollutant source areas and receiving waters to remove pollutants from surface and subsurface runoff (NRCS, 1998). The most prominent pollutant removal processes in filter strips tend to be infiltration of dissolved pollutants and deposition of sediment bound pollutants (Medez-Delgado, 1996). Buffers are used for the treatment of surface run-off from cropland or confined animal facilities. Robinson *et al.* (1996) observed that a 3m wide buffer effectively removed up to 70% of the sediment load from cropland runoff. Edwards (1997) reported that buffers were effective for removing metals found in runoff from fields treated with poultry litter. Barone (1998) reported that buffers were effective for removing nutrients, bacteria and pesticides from surface runoff.

Dillaha *et al.* (1986) has observed that the effectiveness of buffers tended to decrease with time. With proper maintenance, buffers are expected to function for up to 10 years (Dillaha and Hayes, 1991). Sediments accumulate in the buffer over time, large flows from extreme precipitation events may flush the buffer of its sediment load. Without harvesting the biomass grown in the buffer, the trapped nutrients will accumulate, thus increasing the risk of groundwater pollution or increasing the nutrient concentration of waters leaving the buffer.

Cover crops are a source reduction managerial practice. The main purpose of cover crops is to provide soil cover and protection against soil erosion. Cover crops also sequester nutrients over the winter, prevent nutrient loss, and provide a ‘green’ manure source in the spring (Wyland, 1996) if the cover crop is left in field or ploughed under before planting of the primary crop. Another benefit of cover crops is soil moisture management by reducing soil evaporation when plants are dormant (Ewing *et al.*, 1991).

Another valuable natural resource is wetlands. Constructed wetlands have been used to treat municipal waste, industrial and more recently agricultural wastes (Reed, 1991). Wetlands are cost effective, efficient and suitable method for treating a wide range of pollutants. Magmedov and Yakovleva (1988) found that sulphates, ammonium and nitrate concentrations of up to 100 mg l^{-1}, chlorides up to 1500 mg l^{-1} and suspended solids up to 300 mg l^{-1} did not suppress wetland biocenosis. Due to the complexity of wetland systems, wetland behaviour will vary from site to site (Reed and Brown, 1992).

Nutrient management plans are one of the most common ways to address diffuse pollution from agricultural lands (NRCS, 1998). Nutrient management is a source reduction managerial practice and aims to enhance crop yields while minimising the loss of nutrients to surface and groundwater resources. This is achieved by managing the amount, form, placement and timing of plant nutrient applications (Beegle and Lanyon, 1994). In most cases nutrient management plans are based on the nitrogen needs of the crops. When the amount of fertiliser applied to cropland is based on crop nitrogen needs over application of phosphorus may occur because the nitrogen content of fertilisers are generally less than the phosphorus needed by crops (Sharpley, 1994).

In the past it was assumed that the excess phosphorus would be held by soil minerals and not be available for transport (Sims, 1995). The over application in some areas has resulted in the phosphorus saturation of agricultural soils. Therefore any phosphorus applied to these soils would increase the potential for degradation of the aquatic habitat in the receiving waters. This is especially true for orthophosphorus, which is highly mobile by surface runoff and is an essential nutrient in the eutrophication process (Sharpley, 1994).

Effective nutrient management planning and effective execution of best management practices requires a thorough understanding of the behaviour of nitrogen and phosphorus in the system and of the controls of the losses. This understanding is still incomplete (Oenema & Roest, 1998).

2.3 Possible Future Scenarios

Scenarios are used to determine how conditions may change in the future. A scenario is a coherent, internally consistent and plausible description of a possible future state of the world (Parry and Carter, 1998). A scenario is not a prediction of the future, since use of the term “prediction” or “forecast” implies that a particular outcome is most likely to occur. Rather, a scenario represents one of any number of possible futures, which can be used to provide data for vulnerability, impacts and adaptation studies; to scope the range of plausible futures; to guide and explore the implications of adaptation and mitigation decisions; and to raise awareness of climate change issues. They provide a range of possible futures that allow consideration of the uncertainty relating to the different pathways that exist for future social, economic and environmental change.

2.3.1 Climate Change

The Earth's climate has been changing throughout its history and until now this has been mostly due to natural causes (UKCIP, 2002). According to the Intergovernmental Panel on Climate Change (IPCC) the increase in surface temperature over the 20th century for the Northern Hemisphere is likely to have been greater than that for any other century in the last thousand years (IPCC, 2001).

Climate changes affect the hydrological cycle, thus modifying the transformation and transport characteristics of nutrients. At the current stage of knowledge, large-scale global circulation models (GCM) are probably the best available tools to estimate the effects of increases in greenhouse gases on rainfall and temperature patterns through continuous three dimensional simulation of atmospheric, oceanic and cryospheric processes (Bouraoui *et al*., 2002).There is a general consensus that the Earth will be subject to warming (Nijssen *et al*., 2001). For Europe, Beniston and Tol (1998) reported a surface air temperature increase by 0.8^{o}C during the 20th century with large temporal and spatial variations.

Four GCMs have been reviewed thoroughly by the IPCC (IPCC, 1996): CSIRO-Mk2 (Hirst *et al*., 1996), ECHAM4 (Roeckner *et al*. 1996), CGCM1 (Flato *et al*., 1999) and HadCM2 (Johns *et al*., 1997). GCM predictions of temperature increases are often associated with large uncertainties, the source of which is well documented in the literature (Allen *et al*., 2000; Mitchell & Hulme, 1999; Reilly *et al*., 2001; Visser *et al*., 2000). Hulme and Carter (1999) summarise that these uncertainties stem from the coarse resolution of the models and their representation of atmospheric and other processes.

Despite these uncertainties, monitoring, research and model simulation results show that climate change can have a significant impact on soil and water resources (Murdoch *et al*., 2000). Reviews on the impact of climate change on the water cycle are available for the US (Gleick, 1999) and for Europe (Arnell, 1999). In its latest report the IPCC (IPCC, 2001), notes that long term studies have already shown the adverse effects of increased temperature on physical and biological systems in many parts of the world. Murdoch *et al.* (2000) reviewed the potential impact of climate change on surface water quality in North America. It was noted that an increase in diffuse source pollutant loads is among the effects to be expected. As seen earlier, diffuse losses of nutrients, especially those from agricultural origins, are among the major contributors to the total load of nutrients to the river system (Novotny & Olem, 1994).

Kallio *et al.* (1997) looked at the impact of climatic change on agricultural nutrient losses in Finland. Changes in nitrate and particulate phosphorus losses from agricultural areas were estimated in a new equilibrium climate assuming an increase of 4.7^{o}C in temperature and 12% in precipitation as compared to the present climate. On the basis of model estimations Kallio *et al.* (1997) predicted increases in precipitation and temperature would increase the nitrogen loss from agricultural areas to surface waters. The main reasons for the increase were thought to be the acceleration of organic matter mineralization in agricultural soils and the increased water flow through the soil column. Particulate phosphorus losses showed a mean decrease for all agricultural land in Finland. The main causes for the decrease were attributed to the shorter period of frozen soil and the reduced snowfall, both of which reduce surface run-off.

The predicted reduction in particulate phosphorus in Finland due to climate change may explain the conclusions drawn by Frisk *et al.* (1997). By looking at climate change and lake eutrophication with the use of a dynamic simulation model they concluded that an increase in the spring peak of phytoplankton could be expected, but the average biomass would remain the same. According to the simulations, the effects of climate change on the trophic status of lakes would not be great. In contrast Hassan *et al.* (1998) predicted that an increase in phytoplankton growth rate would occur throughout the year due to climate change using a hydro-quality mathematical model and future climate estimates from HadCM2SUL. On the basis of modelling results it can be concluded that climate change may cause an increased risk for eutrophication in some lakes.

2.3.2 Agriculture Futures

The consequences of global climate change on agriculture and ecosystems are highly uncertain (Clifford *et al*., 1996). There will be both positive and negative effects for agriculture. For example if global warming causes a change in climate zones to higher latitudes then parts of Russia and Eastern Europe will become wetter and as a result their agricultural productivity would rise. The arable areas of North America and Southern Europe would become drier and warmer; as a result there will be more heat waves and droughts, which could mean widespread crop failures. This could lead to worldwide famine (Liverman, 1986).

The impact of climate change on arable crops, horticulture, weeds, pests and diseases, grasslands and livestock includes changes in the location of agricultural activities, earlier development and growth, changed yields and quality (Table 2.3.1). In the UK it is predicted for winter wheat that a temperature increase of 2ºC and a precipitation decrease of 10% will lead to increases in cereal production in western England, whilst areas of East Anglia become less suitable because of drought. Brignall *et al.* (1994) suggested that sunflower and grain maize might become more common in areas currently dominated by cereals.

Table 2.3.1: Predicted effects of climate change on agriculture over the next 50 years (Leake, 2000)

Climatic element	Expected changes by 2050's	Confidence in prediction	Effects on agriculture
CO_2	Increase from 360 ppm to 450 - 600 ppm	Very high	Good for crops: increased photosynthesis; reduced water use
Sea level rise	Rise by 10 -15 cm Increased in south and offset in north by natural subsidence/rebound	Very high	Loss of land, coastal erosion, flooding, salinisation of groundwater
Temperature	Rise by 1-2°C. Winters warming more than summers. Increased frequency of heat waves	High	Faster, shorter, earlier growing seasons, range moving north and to higher altitudes, heat stress risk, increased evapotranspiration
Precipitation	Seasonal changes by ± 10%	Low	Impacts on drought risk soil workability, water logging irrigation supply, transpiration
Storminess	Increased wind speeds, especially in north. More intense rainfall events.	Very low	Soil erosion, reduced infiltration of rainfall
Variability	Increases across most climatic variables. Predictions uncertain	Very low	Changing risk of damaging events (heat waves, frost, droughts, floods) which affect crops and timing of farm operations

As well as climate the main drivers that will shape agriculture in England and Wales under the possible futures are (Berkhout *et al.*, 1999):

- Change in or abandonment of EU agricultural policy (especially CAP reform)
- Demand for and supply of agricultural commodities in England and Wales and world markets due to population growth, economic prosperity and preferences
- Pressures on natural resources
- Importance given to social and environmental issues
- Technology development.

Morris (2003) has constructed future agricultural and related environmental scenarios based on the above drivers, drawing on the methodology developed by the UK Foresight Programme (DTI: 1999, 2002) (Table 2.3.2). Foresight considers long term futures and possible implications for UK industry and society. The programme constructed four possible futures, which are distinguished in terms of social values and governance (Berkhout *et al.*, 1998).

Table 2.3.2: Future Agricultural and Related Environmental Scenarios linked with the Foresight Programme (Morris, 2003)

Foresight Scenario	Agricultural Policy Scenario	Intervention regime
	Baseline	Moderate: Existing price support, export subsidies, with selected agri-environment schemes
World Markets	World Agricultural Markets (without CAP)	Zero: Free trade: no intervention
Global sustainability	Global Sustainable Agriculture (Reformed CAP)	Low: Market orientation with targeted sustainability ‘compliance’ requirements and programmes
Regional enterprise	Regional Agricultural Markets (Similar to pre-reform CAP)	Moderate to High: price support and protection to serve national and local priorities for self sufficiency, limited environmental concern
Local Stewardship	Local Community Agriculture	High: locally defined support schemes reflecting local priorities for food production, incomes and environment

The four socio-economic scenarios described above form an important part of possible agricultural futures. Socio-economic change has a major effect upon the vulnerability of agriculture, water and biodiversity to climate change (Shackley & Wood, 2001). As with climate change scenarios, socio-economic scenarios help to provide indicative measures of change, reducing a multiplicity of possibilities to more manageable proportions. Parry *et al.* (1998), for instance, has shown that trade liberalisation is likely to have a greater effect upon agriculture in the UK over the course of the next few decades than climate change.

Future land use under different climate change scenarios and socio-economic scenarios have been modelled as part of a ‘Regional climate change impact and response study’ (RegIS). This has been carried out to adapt and develop mathematical climate change impact models for the coastal, agriculture, water and biodiversity sectors, in two regions; East Anglia and North West England (Holman & Rounsevell, 2001). Five scenarios have been modelled; the UKCIP98 2050’s Low and High climate scenarios with no socio-economic change and then with linked socio-economic (Regional Enterprise and Global Sustainability) and climate scenarios for the 2050’s (Audsley et al., 2001).

The simulations showed that when the socio-economic scenario is unchanged, the proportions of the major crops are also little changed. The area of grassland in the East Anglian region has increased to almost double its current area under the 2050's high scenario, although it is still a low proportion of the total. The majority of this increase is due to land changing from arable to pastoral agriculture due to the risk of flooding. When the Global Sustainability socio-economic scenarios are introduced break crops such as oilseed rape, beans and peas are increased at the expense of cereals and the areas of sugar beet and potatoes decrease. The effect of the Regional Enterprise socio-economic scenario is to eliminate all break crops other than sugar beet and potatoes, but these do not increase. Instead barley and oats increase and wheat reduces.

In terms of nutrient input RegIS predicts that nitrogen requirement is almost unchanged for all the scenarios except for the Regional Enterprise socio-economic scenario where the large increase in yields of the crops means there are correspondingly large increases in the economically optimum level of nitrogen to be applied. Combined with the change in cropping this amounted to an increase of over 60% in nitrogen requirement for the region.

The results from this study show that it is possible to integrate agricultural land use modelling directly into climate change studies involving socio-economic scenarios and river modelling. It has also highlighted the need under some scenarios for agricultural adaptations to be undertaken in response to climate change.

The Centre for Rural Economic Research (University of Cambridge) (2004) has looked at the future of agriculture over a shorter timescale than RegIS. It aims to produce estimates of agricultural land use in England and Wales by 2015, the time when the EU Water Framework Directive comes into force. Table 2.3.3 shows projections for change in agricultural activities by 2015 in the Eastern Region of the UK.

Table 2.3.3: Projections for Change in Agricultural Activities by 2015 in the Eastern Region (The Centre for Rural Economic Research, 2004)

Activity	Projected Change (% to 2015)
Total Agricultural Area	-0.9
Grass	-10.0
Wheat	11.5
Winter Barley	6.0
Spring Barley	12.0
Oats	12.0
Potatoes	-8.0
Sugar Beet	-30.0
Field Beans	15.0
Peas (harvested dry)	25.0
Oilseed Rape	12.0
Linseed	-50.0
Maize	10.0

Unlike RegIS a decrease in grassland has been predicted, however predictions for break crops for both projects show a general increase except for sugar beet and potatoes.

The importance of farm and state level adaptations to climate change and variability has been demonstrated in further studies (e.g. Rosenberg *et al.* 1989; Waggoner 1983; White, 1974). Adaptations to climate change exist at the various levels of agricultural organization. In temperate regions, farm-level adaptations include changes in planting and harvest dates, tillage and rotation practices, substitution of crop varieties or species more appropriate to the changing climate regime, increased fertilizer or pesticide applications, and improved irrigation and drainage systems. Governments can facilitate adaptation to climate change through water development projects, agricultural extension activities, incentives, subsidies, regulations, and provision of insurance.

2.4 Modelling of Non-point Source Pollutant Loads

In ideal conditions it would be possible to continuously measure tributary flows and the pollutants carried in the flowing water. Under such conditions, the product of these two variables would provide an extremely accurate measurement of pollutant loads. In reality, however, the available field, technical and financial resources are never adequate to accurately measure a pollutant load from all potential non-point sources in a river basin. The result is that individuals and agencies often must make use of mathematical models to estimate and/or predict present and future non-point source pollutant loads to water bodies (Jorgensen *et al.*, 1996).

Models can be viewed as physical or theoretical tools that attempt to provide a ‘picture’ of the real world, without the necessity of having to actually ‘construct’ a representation of the real world. The development of mathematical modelling techniques (chemical, physical, or biological) has not yet reached a stage where modelling is a completely accurate and reliable basis for quantitatively describing and/or forecasting non-point source pollution processes (Jolankai *et al*., 1999). In addition, even if modelling techniques were firmly established, a lack of necessary data would still limit the practical application of sophisticated modelling techniques. Nevertheless, in order to design realistic non-point source pollutant control strategies, relevant cause and effect relationships must be quantified. Pollutant inputs from all major sources must be related quantitatively to their measured impacts in the components of the environment receiving the pollutants.

2.4.1 Modelling Approaches

River basin models can be classified into three categories: empirical, physically based and conceptual models.

Empirical models do not utilise physical laws to relate input to output. Conducting measurements on both inputs and outputs develops them. In empirical models, the variation within the river basin characteristics is not accounted for directly. However, the mathematical formulae used implicitly represent the physical system within the range of data they were developed from. SWATCATCH (Hollis *et al.,* 1996), HSPF

(Crawford and Linsley, 1966), INCA (Wade *et al.*, 2002) and INCA-P (Wade *et al.*, 1988) are some examples of empirical models. Since this type of model is not based on physical characteristics of the river basin, empirical models cannot be generalised to other locations and scenarios without reducing their accuracy (Wood and O'Connell, 1985). These models are generally only applied to conditions for which the parameters have been calibrated.

Physically based models are those based on complex physical laws and theories. The 'real world' is simplified to different degrees in space and time. Physically based models are intended to represent a synthesis of the individual components, which affect the river basin, including the complex interactions between various factors and their spatial and temporal variability. Physical modelling of a river basin would imply using fundamental physical equations at a small scale, such as the law of conservation of mass. Given the complexity of a river basin, this can be done in practice only for water routing (Ouarda *et al.*, 1997). Consequently, only short term flow forecasts can be obtained from physically based models, since the effects of precipitation, infiltration and evaporation must be negligible.

Few physically based river basin models have been used in practice as they are over complicated for the level of data often available (Brooks *et al.*, 2003), although many have been developed and tested as part of research projects (ANSWERS; Beasley *et al., 1985* and MIKE-SHE; DHI, 1998). Extensive data are needed for even the simpler models and just as much calibration is needed as with the use of other model types.

Conceptual models fall in a category between physically based models and empirical models. They usually represent physical formulae in a simplified form. Singh (1988) noted that these models are able to provide useful results efficiently and economically for some problems. Conceptual models can either be lumped or distributed. Lumped models treat the river basin as a single unit, with state variables that represent the averages over the river basin area. Distributed models make predictions that are distributed in space, by splitting the river basin into a large number of elements or grid squares. These models are capable of reflecting changes in river basin characteristics if the parameters used are physically based. Conceptual models are therefore, very useful for inferring the distribution, magnitude, past, present and future behaviour of processes with limited observations.

All three model types have their advantages and disadvantages, but conceptual modelling allows future river basin behaviour to be modelled and requires less extensive data than physically based model types. Of the hundreds of conceptual models on the market the Soil Water Assessment Tool (SWAT; Arnold *et al.*, 1996) (see below) is designed for looking at long term outcomes. This makes it very suitable for looking at future climate and land use change within a river basin.

2.4.2 Soil Water Assessment Tool

The Soil Water Assessment Tool (SWAT) was created by the United States Department of Agriculture (USDA) as an integrator of the simulators CREAMS (Knisel, 1980), GLEAMS (Leonard *et al.*, 1987), SWRRB (William and Nicks, 1994) and ROTO (Neitsch *et al*., 2000). The model was developed with the object of modelling the effect of agricultural practices in large un-gauged basins (Arnold *et al*. 1994; Srinivasan & Arnold, 1994). It is a spatially distributed, river basin level biophysical model, whose components include; weather, surface run-off, return flow, percolation, crop growth, irrigation, groundwater flow, reach routing and nutrient and pesticide loading among other features (Srinivasan *et al*., 1995). It operates on a daily time step and is designed to study long-term impacts (Neitsch *et al*. 1999).

The model itself is based on the water balance equation. Surface run-off is calculated applying an improved SCS (Soil Conservation Service) Curve Number approach (Arnold *et al*., 1998). The percolation component consists of a linear storage with up to ten layers. The flow rate is governed by the hydraulic conductivity and the available water capacity of each layer. For lateral subsurface flow, a kinematic storage model is used. Percolation from the root zone recharges a shallow aquifer (Arnold *et al*., 1993), which is also connected to stream flow. The model calculates evaporation from soil and transpiration from plants separately, as described by Ritchie (1972). The actual evaporation is a function of soil water content, plant type and soil depth.

Transpiration is computed as a linear function of potential plant evapotranspiration and leaf area index. Canopy storage for each crop is also included.

SWAT also contains a plant growth model, which is a simplified version of the plant growth approach of the EPIC model (Williams *et al*., 1983). It is based on accumulating heat units, harvest index for the partitioning of grain yield, the Monteith approach for potential biomass (Monteith, 1977) and water, nutrient and temperature stress concepts. A single model is used for simulating all the crops considered. Tillage systems and agricultural management can be specified for each crop.

2.4.3 SWAT Applications

Natural river basin systems maintain a balance between precipitation, runoff, infiltration, and water, which either evaporates from bare soil and open water surfaces or evapotranspires from vegetated surfaces, completing the natural cycle. The understanding of this hydrologic cycle at a river basin scale, and the fate and transport of nutrients, pesticides and other chemicals affecting water quality is essential for development and implementation of appropriate river basin management policies and procedures (Montanari & Uhlenbrook, 2004).

In recent years, application of models has become an indispensable tool for the understanding of the natural processes occurring at the river basin scale (Downs, 2001). As the natural processes become more modified by human activities, application of integrated modelling to account for the interaction of practices such as agricultural management, water removals from surface bodies and groundwater, release of sewage effluent into surface and sub-surface waters, urbanization, etc., has become more essential (Slater *et al*. 1993).

The program SWAT due to its continuous time scale, distributed spatial handling of parameters and integration of multiple processes such as climate, hydrology, nutrients and pesticides, erosion, land cover, management practices, channel processes, and processes in water bodies has become an important tool for river basin-scale studies (Cau *et al.,* 2003; Conan & Bouraou, 2003; Griensven *et al*., 2001; Kirsch *et al*., 2002; Van Liew & Garbrecht, 2003).

SWAT has been applied to numerous projects and applications have shown some promising results. Peterson and Hamlett (1998) applied SWAT to model the hydrological response of the Ariel Creek river basin of northeastern Pennsylvania, which contains fragipan soils and wetlands. The report revealed that model calibration yielded Nash-Sutcliffe coefficients (E_{NS}) of 0.04 and 0.14 when comparing daily and monthly flows respectively. These E_{NS} values are particularly low, optimum values should be as close to 1 as possible. Eckhardt and Arnold (2001) used a stochastic global optimization algorithm to perform the automatic calibration of SWAT simulation on a low mountain range river basin in central Germany. The results indicated a good agreement of measured and simulated daily flow with an E_{NS} value of 0.7. They concluded that the mean annual stream flow is slightly underestimated by 4%.

The applicability of SWAT for estimation of agricultural nutrient loads and the effects of agricultural management practices on these loads is well documented. Saleh *et al.* (2000) applied SWAT to assess the effect of dairy production on water quality within the Upper North Bosque river basin in north central Texas. Model outputs were compared to flow, sediment and nutrient measurements for 11 stream sites within the river basin for the period 1993 – 1995. Results indicated that SWAT was able to predict the average monthly flow, sediment and nutrient loadings at 11 stream sites reasonably well with E_{NS} values ranging from 0.65 to 0.99.

Outside of the USA Melo de Souza *et al.* (2003) applied SWAT to a small river basin of 1.8 hectares at Darnum in West Gippsland, Victoria, Australia. The river basin is in a typical rural area and has been monitored since 1994. The main objective of the project was to assess the total phosphorus (dissolved and particulate) concentration in overland flow – run-off – from farmland to the water body. In terms of run-off, outcomes from model simulations were not accurate when compared with the measured data. The model over predicted run-off, giving an E_{NS} of 0.48. This was attributed to the model being based on the Soil Conservation Service (SCS) run-off equation, which was developed in the 1950's for estimating the run-off yield from rainfall for a variety of soil types and land use conditions of river basins in the US. The SCS curve number (CN) is a function of soil permeability, land use and antecedent soil water conditions (Neitsch *et al.*, 2000). Some adjustments in the CN curve number and the soil water capacity value were necessary to refine the model performance.

The total phosphorus concentrations needed fewer adjustments than the run-off simulations and E_{NS} of 0.99 was obtained. The initially poor predictions by the model could be related to default values, based on current USA characteristics that were likely to be different in Australia. It is also possible that the differences were related to the size of the river basin, which was smaller than those for which the model has been calibrated in the USA.

The suitability of SWAT for the use on smaller river basins in the UK has been demonstrated in a number of projects. The SWAT model has been applied using data from a Unilever experimental river basin at Colworth, (1.415 km^2) Bedfordshire, UK (Kannan, 2004). This is an intensively monitored river basin and provided perhaps the best spatial and temporal resolution data required for a SWAT model run. The inconvenience with using such a data set is that SWAT was designed as a management model and as such is best suited to application at the larger river basin scale. Nonetheless, calibration and validation carried out for daily data gave E_{NS} values of between 0.63 and 0.70. The work carried out at Colworth therefore provided the opportunity to investigate the operation of the SWAT software in considerable detail, providing promising results.

Under the Terrestrial Run-off Modelling for Risk Assessment of Chemical Exposure (TERRACE) project the Exe river basin in south-west England has been modelled using SWAT (White *et al.*, 2003). The overall aim of the TERRACE project was to develop a simulation model for evaluation of diffuse source chemical run-off at the regional scale across Europe. The purpose of modelling the Exe river basin was to demonstrate how SWAT software could be used to provide contaminant inputs to the GREAT-ER model. The model was calibrated at three sites and gave E_{NS} values of 0.35, 0.67 and 0.40 respectively. Although these results are slightly lower than other modelled river basins in the UK the final predictions by SWAT showed its flexibility to configure a river basin in an environment outside the USA.

In the UK SWAT has also been evaluated to determine its suitability of use in the Ythan river basin under a benchmark criteria to assess appropriateness of models for use in implementation of the Water Framework Directive (Dilks *et al*., 2003). The Ythan river basin is an area where diffuse nutrient pollution, specifically nitrates from agricultural fertiliser, has been identified as the main pressure (Edwards *et al*., 2003). SWAT performed successfully when evaluated against qualitative diffuse pollution benchmark criteria. It was therefore considered to be suitable for assessing the long-term implications of various management strategies of relevance to the Ythan river basin. These included good agricultural practice (e.g. timing and amount of fertiliser applications), the effects of buffer strips and land use change.

Further evaluation of SWAT as a tool for identifying potential source areas of water, sediment, nitrate and phosphate with the Water Framework Directive in mind has been carried out in the Wensum river basin in East Anglia (White *et al.*, 2004). Two management scenarios have been run using different crop rotations and tillage patterns to demonstrate the sensitivity of the SWAT model to the types of change that might be envisaged as management controls on sediment and nutrient loss. Calibration results gave an E_{NS} of 0.93 for predicted flows. Nutrient results have also been successfully modelled and demonstrated much more variability in nutrient transport than can that monitored using a four-weekly sampling exercise, such as that currently in use by the Environment Agency.

SWAT has been successfully applied in Europe and more specifically the UK. The most recent published work conducted within the UK has however, focused on evaluating the impact of climate change on water quality. Bouraoui *et al.* (2002) applied SWAT to the Yorkshire Ouse river basin to assess the impact of potential climate change on nutrient loads to surface water. The study followed the impact approach described by Carter *et al.* (1994) using a three-step procedure: calibrate and validate the hydro-geochemical model using measured climate data, define the climate change scenarios and perturbation to be applied to the actual climate data, and then run the model using the baseline and perturbed climate time series.

Climate scenarios were developed from the output of climatic Global Climate Models (ECHAM4, CSIRO-Mk2, CGCM1 and HadCM2) experiments. For each of the climate scenarios, two variables were used: precipitation and temperature. The perturbations of the baseline daily time series were carried out simply by adding the estimated change in temperature in each month to the baseline of everyday of that month. Similarly, the baseline daily precipitation of each month was multiplied by the estimated change in precipitation in that month to obtain the daily perturbation.

All climate scenarios, except one, predicted an increase in surface water flow and all climate scenarios significantly affected not only water quality and nutrient loads from agricultural areas but also crop growth patterns. This agrees with other studies (Hanatty & Stefan, 1998; Kallio *et al*., 1997). One of the major conclusions is that climate change will increase the nutrient losses to surface water firstly by accelerating soil processes such as mineralization of organic matter and by increasing the amount of water transiting through the soil profile to the river network. Furthermore all scenarios predicted a shift in crop growth. This will affect the soil and crop management, so that traditional crop rotations and management practices will have to be adjusted.

Literature has shown that SWAT is responsive to both management practices and climate; however the choice of future management practices will be greatly influenced by predicted future climate change impacts. It is therefore judged that SWAT can be used to study the impact of future climate and land use scenarios to predict the impacts of these scenarios on flow and nutrient dynamics in the Broads.

Chapter Three **Study Area and Available Data**

3.0 Spatial Data

3.0.1 Geography/Location

The Broads are a group of small, very shallow lakes, interconnected by a tidal river system in eastern England. Geographically they extend over an area of approximately 560 km^2 in the lower valleys of the rivers Waveney, Yare and Bure together with two Bure tributaries the Ant and Thurne (Bennion *et al*., 2001).

The Upper Thurne broads are located approximately 5km southeast of Stalham, forming part of the headwaters of the River Thurne, approximately 4km from the coast (Fig 3.0.1).

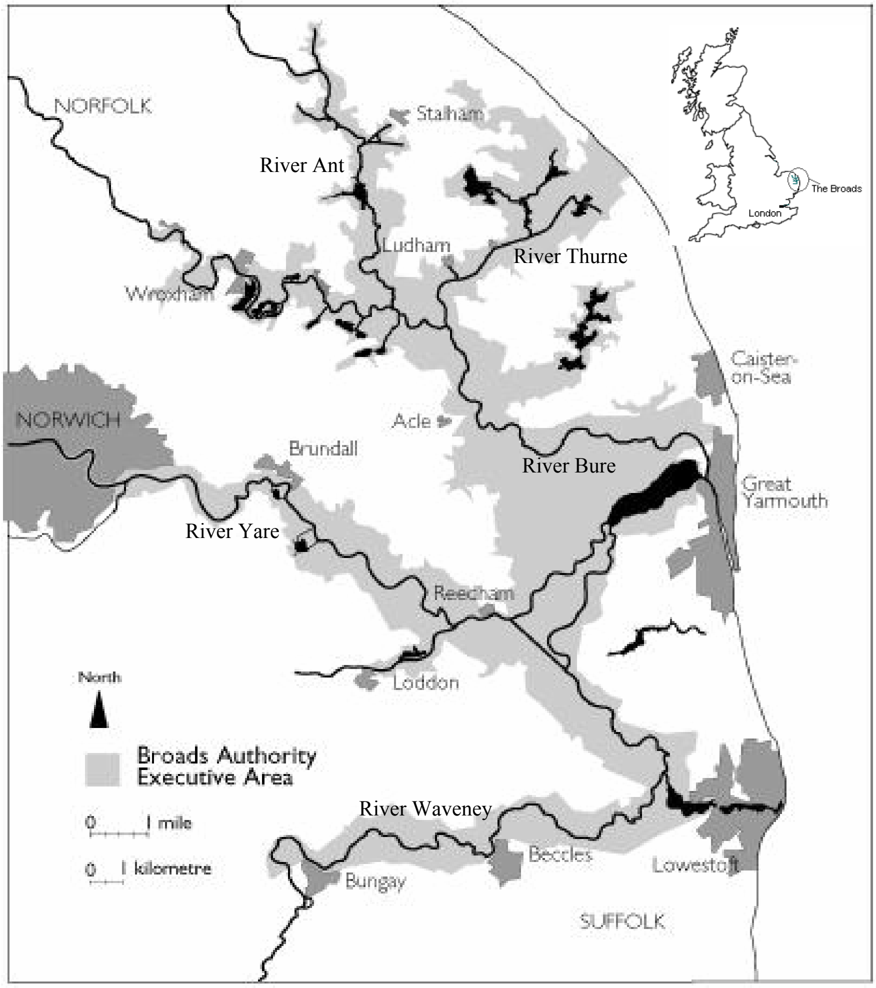


Figure 3.0.1: Map of the Broads

3.0.2 Topography

The relief of the Thurne river basin is subtle, with some distinct topographical features. There are two areas of higher ground, ranging from +16mOD in the northwestern parts around Hickling to +23mOD along the southern watershed near Martham. There is a minimum of -2mOD in the deeper drained levels. Slightly higher areas within the broads are called ‘Holmes’.

A 30 km belt of sand dunes with heights up to 10mOD and a width of approximately 100 m protect low lying areas from flooding by the sea (Holman & Hiscock, 1998).

3.0.3 Geology

Figure 3.0.2 shows a geological cross section of the Upper Thurne Broads. It can be seen that the study area falls within the complex drift deposits of the Breydon Formation. These deposits are comprised of peat, clay, silt and sand, which are highly variable both laterally and vertically. In areas of peat diggings the Breydon Formation may be very thin or non-existent. It is believed possible that the base of Hickling Broad (covered by recent fluvial sediment) consists of sands and gravels of the Pleistocene drift formation (Power *et al.* 2001).

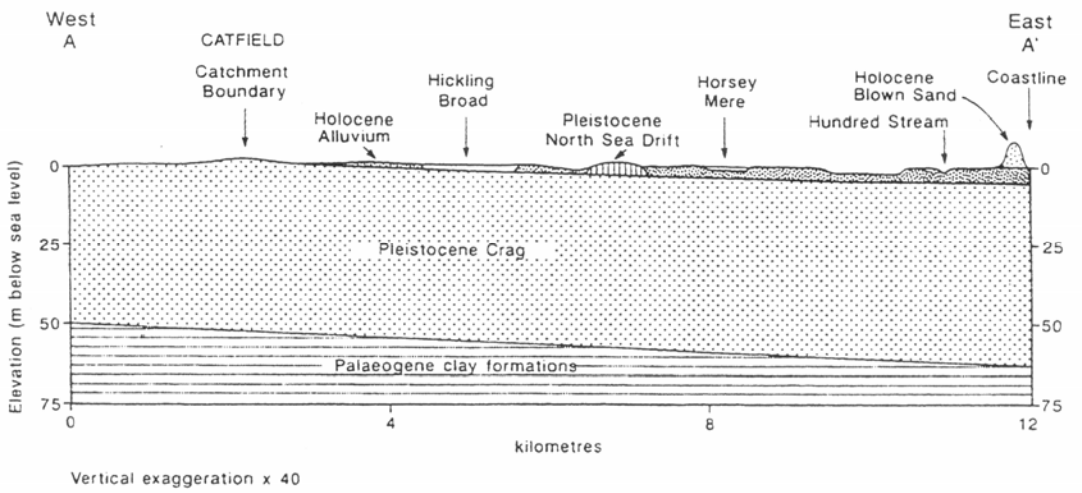


Figure 3.0.2: General geological section across the River Thurne river basin (Holman & Hiscock, 1998)

Underlying the drift are rocks of the Pleistocene Crag Group. The Crag consists of marine shelly sands and silty and micaceous clays. Middle to late Pleistocene deposits consist of glacial and interglacial deposits including the Kesgrave Sands, Cromer Forest Bed and Anglian glacial till deposits (Holman *et al*., 1999).

Underlying the Crag are beds of Palaeogene Age, namely the London and Thanet Formations. These clays are mainly mudstones and siltstones with ash layers and are underlain by Cretaceous Chalk.

3.0.3 Soils

The higher areas in the river basin have coarse soils such as Gresham (0711v) and Wick (0541s) Associations. Both are non-calcareous and therefore have potential to become acid from leaching. The level of acidity is only slight except around the edges of the lowest ground (Burton, 1990). The main river valley contains a ground gley soil such as the Hanworth (0871c) Associations, with the large side valleys comprising of peat soils such as the Altcar 2 (1022b) Association. There are also sandy soils present in the river basin bordering the dunes, which belong to the Sandwich (0361) Association (Fig 3.0.3).

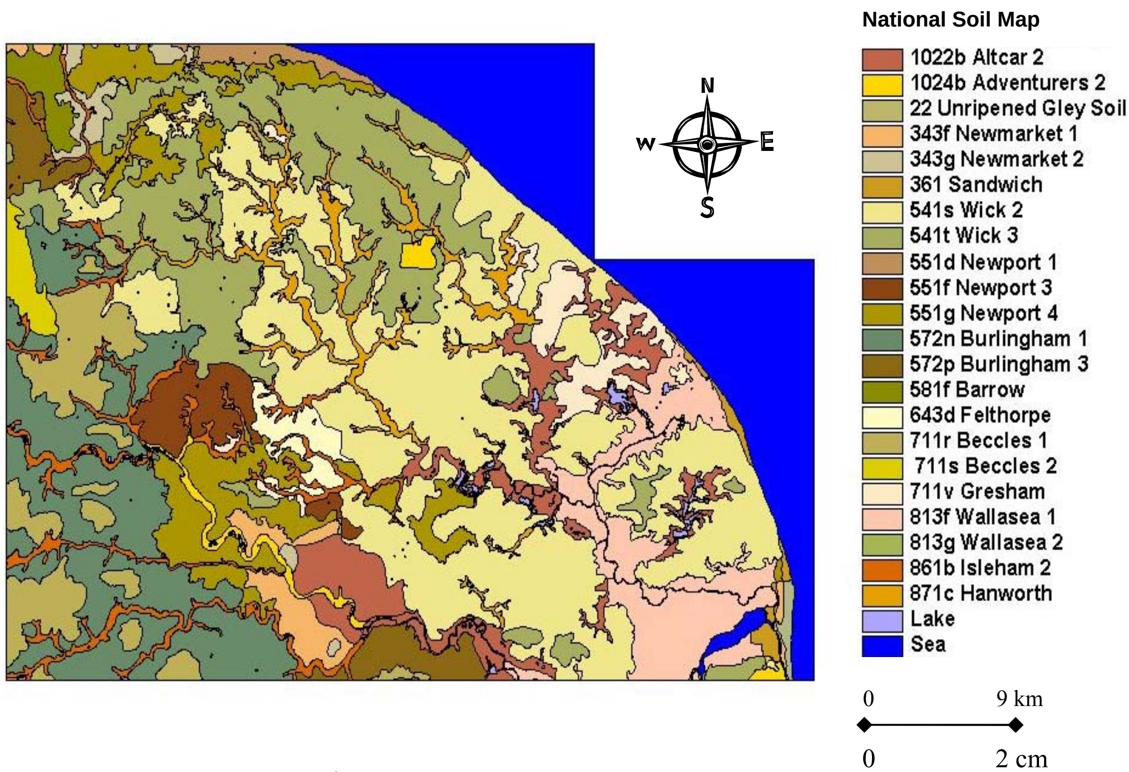


Figure 3.0.3: National soils map of the Broads

3.1 Spatially and Temporally Varying Data

3.1.1 Climate

In common with other parts of East Anglia, the Broads have a slightly more continental climate than other parts of the UK with an annual maximum mean temperature of 12.8°C and minimum temperature of 7.2°C (1961 – 1990). Annual and diurnal temperature ranges are higher than for much of the UK and annual rainfall is lower.

The amount of bright sunshine at Norwich varies from about 50 hours in the months of December and January to nearly 200 hours in May, June and July (George, 1992). However, 'frets' (low level mists) tend to occur on sunny days for up to 4 miles inland, these tend to dissipate further inland as a result of the warming influence of the sun on the land. It is for this reason that sites such as Wroxham can be basking in bright sunshine, when Hickling Broad and Horsey Mere, only a few miles to the east are covered in thick mist.

Annual rainfall is approximately 636.8mm yr^{-1} for the period between 1961 and 1998 (Power *et al*. 2001). Rainfall is unevenly distributed throughout the year, with 60% occurring in the second half of the year. Autumn is the wettest with 31% of the average annual rainfall. Spring and winter are the driest seasons with largely cyclonic rainfall and 21% and 22% respectively of total annual rainfall (Burton, 1990). In terms of water levels in the Broads effective rainfall only arrives in the winter and is a minor source of water compared with river basin inputs.

Future climate change may well have a greater impact on the East of England than in other regions. The following two figures (3.1.1 and 3.1.2) show anticipated changes in annual and seasonal temperature in the East of England and anticipated changes in seasonal precipitation in the region.

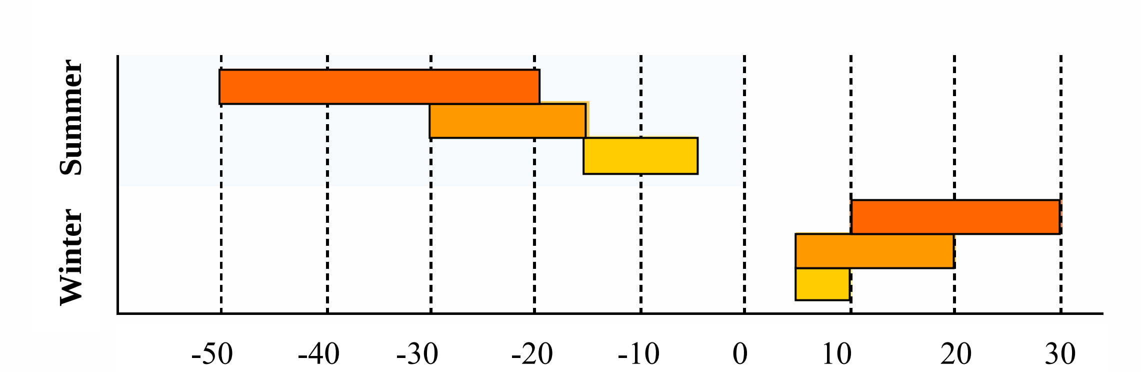


Figure 3.1.1: Changes in East of England annual and seasonal precipitation for the 2020's, 2050's and 2080's (UKCIP, 2002)

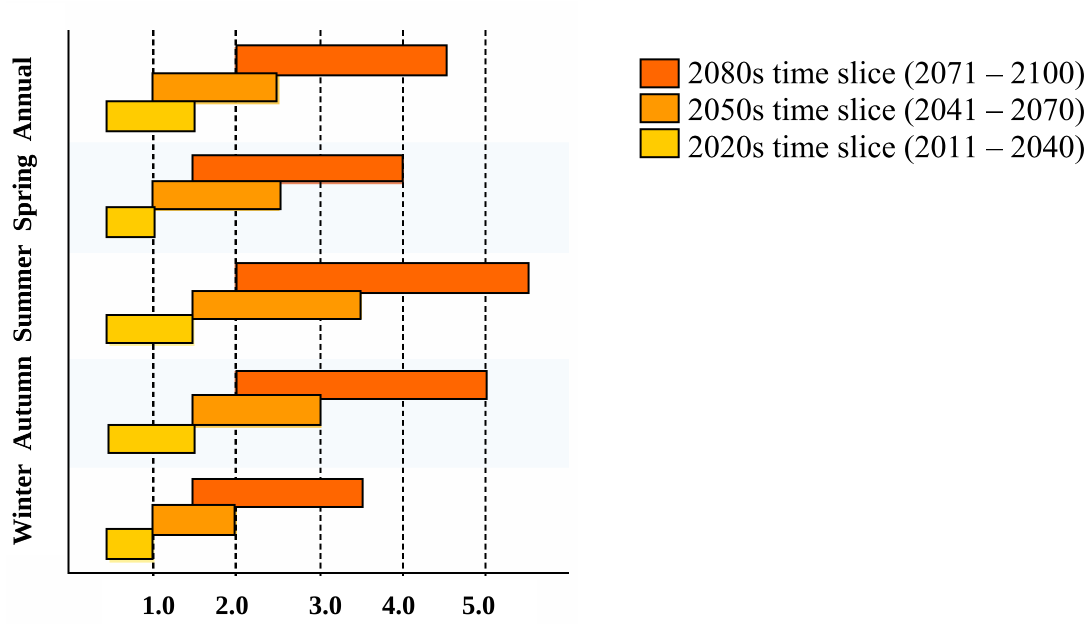


Figure 3.1.2: Changes in East of England annual and seasonal average temperatures for the 2020's, 2050's and 2080's (UKCIP, 2002)

The greater intensity and frequency of winter rainfall may increase the risk of flooding from rivers, while drier summers may put additional pressure on water resources.

The study area has large low lying areas. The region is also sinking very slowly due to geological processes, making it vulnerable to coastal inundation as sea levels rise. Water is already pumped off the land to provide adequate drainage, climate change impacts that are likely to be most significant are increased coastal and fluvial flooding as well as saline intrusion. Increased climatic variability and more intense winter rainfall will put pressure on drainage systems; low summer river flows may affect water quality.

Increased temperatures will result in an increased thermal growing season with reduced summer precipitation and soil moisture greatly affecting agriculture in the region. Adaptation will be required in many farming activities, such as the timing of planting and harvesting, the level and timing of fertiliser applications and ploughing techniques. Adaptation to water pressures will also be very important. Farmers will need to consider growing crops with lower water requirements. These climate impacts will need to be taken into account when looking at development of future scenarios for use in the river basin scale model.

3.1.2 Land Use

Up until the early 19th century the Thurne river basin was poorly drained marshland with arable farming restricted to the Holmes. With better drainage, productivity increased and by the end of the 19th century some of the marshland was under arable crops, the remainder being grazed. Most fields were small, usually less than 5ha. In 1978 – 1981 dyke removal allowed the amalgamation of small fields to large fields preferred by modern arable farmers.

Today with the available incentive schemes in the area from DEFRA as part of the Broads Environmentally Sensitive Area (ESA) there has been an increased uptake in sustainable farming practices with controls on maximum nutrient application rates (Hoare, 2002). Table 3.1.1 shows that the proportion of land covered by permanent grassland and set aside has increased between 1995 and 2001. This was matched by declines in the Thurne area of cereal and other arable production. Arable reversion from crops to permanent grassland and extensification of livestock has therefore been noticeable.

Table 3.1.1: Land use as a percentage of the total agricultural area within the Upper Thurne river basin (Hoare, 2002)

Agricultural Area	1995	1998	2001
Permanent Grassland	31.4	32.2	42.5
Cereal	31.5	36.2	26.7
Other Arable	24.5	24.4	16.4
Rough Grazing	7.8	5.8	5.5
Set Aside	4.8	1.4	8.9

To allow further investigation into land use change the 1990 CEH land cover map (Fig 3.1.3) and Edinburgh Data Library Agricultural Statistics have been obtained from 1969 - 2000. The use of the EDL data will also allow investigation of land use change and subsequent water quality change over time. The 1969 data will provide a baseline against which to compare subsequent change. It is also hoped that it will give an idea of land use and how it affected water quality in pre-phase 3 ecosystems and in some cases phase 2 systems within the Broads.

In 1990 the largest land use type is tilled land (Fig 3.1.3). The EDL data show that cereal crops, field vegetables (peas) and root crops (potatoes and sugar beet) are the dominant crops in the river basin. There is virtually no rough grazing land, which would receive no additional fertiliser inputs. Set-aside land that would also receive no fertiliser application is scattered throughout the river basin. Since this land would have previously been utilised for intensive arable production (condition of entry into the set-aside scheme) these areas may well still contribute to diffuse nutrient export through mineralization of soil nitrogen and phosphorus reserves in saturated soils.

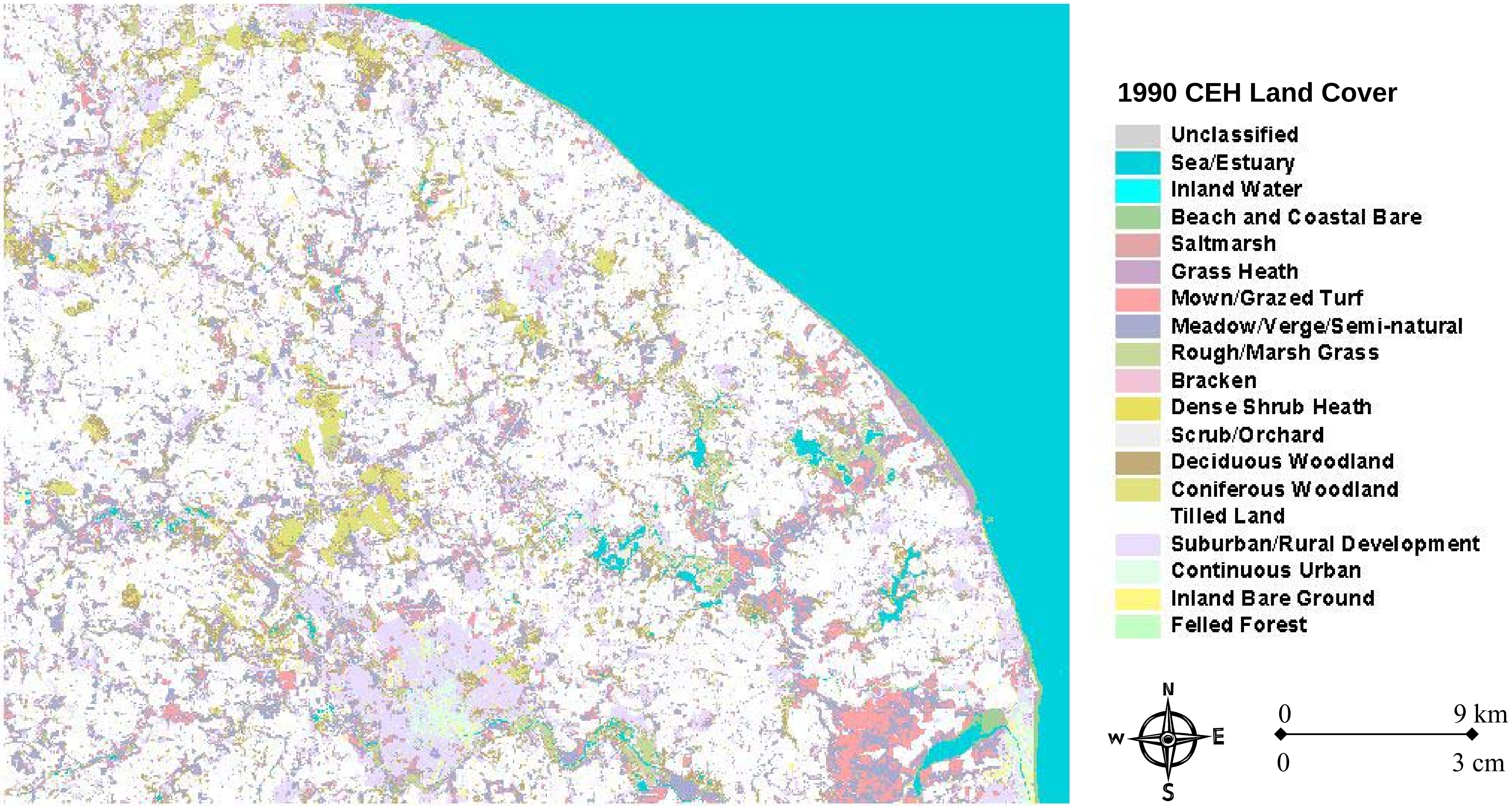


Figure 3.1.3: The CEH 1990 Land Cover Map

Land use change in the UK is characterised by the intensification and expansion of agriculture after the Second World War, dating from the enactment of provisions in the Agricultural Act 1947 (MAFF, 1993). This was designed to increase food production in the UK. 1969 has been selected as a baseline to compare subsequent change, as these are the earliest EDL data available. Earlier data are available from parish summaries of the Annual Agricultural Census returns, which are open for public inspection.

It can be seen from the graph below (Fig 3.1.4) that the area of cereal cropping increased from 1969 – 1976, but then declined as the cultivation of other arable crops such as potatoes and sugar beet increased in response to EC subsidies. As arable cultivation generally increased, the area of permanent and temporary grass has decreased. Crop types requiring high rates of mineral and organic fertiliser application have replaced Land receiving low rates of fertiliser application. This addition of fertiliser will be to land, which is bare for much of the year. Thus, considering historical land use data an increase in nutrient export from diffuse sources might be expected.

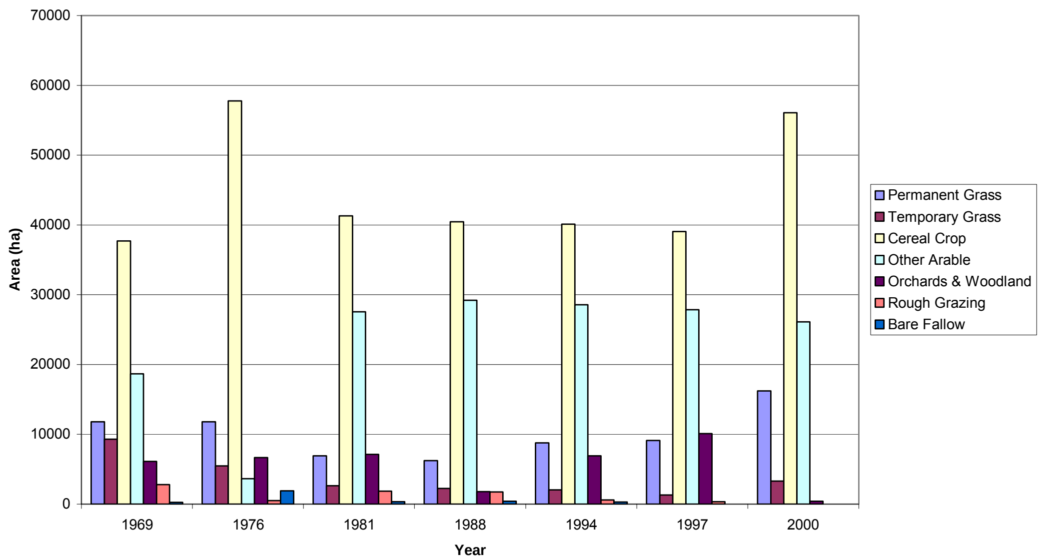


Figure 3.1.4: Land use change in the study area (1969 – 2000)

Under the new EU Water Framework Directive all heavily modified water bodies must reach 'good ecological potential' by 2015. In order to do this in the Broads it has been estimated that approximately 50% of agricultural land needs to be converted to semi-natural vegetation (Moss, 2003). If this is the case then in excess of 43870 ha of land in the study area would need to be taken out of arable production along with the minimisation of cultivation and fertilisation throughout the river basin. This would mean changing over a century's worth of agricultural intensification. It may therefore be more realistic in obtaining a Phase 2 ecological status.

By calculating the annual mean total phosphorus concentration, 35 $\mu g\ l^{-1}$, of four broads (Martham North and South, Blackfleet and Upton) in which phase 2 flora still occurred, and comparing this with estimates of the amount of phosphorus likely to be used by plants, Phillips (1977) suggested that the switch from Phase 2 to 3 occurred once the annual mean concentration of total phosphorus exceed 100 $\mu g\ l^{-1}$. Subsequent studies by Moss and others lend general support to this hypothesis. The switch to Phase 3 occurred in most Broads in the early 1950's, but the larger Thurne Broads retained Phase 1/2 flora similar to that still found in Martham and Blackfleet Broads until the late 1960's (George, 1992).

Habitat features found in the Thurne river basin (other than marshes and dykes) include open water (broads); river; reed bed; fen and alder carr (Fig 3.1.5). The following sites have also been designated (Fig 3.1.6):

Upper Thurne Broads and Marshes:	SSSI, including Hickling, Horsey Mere, Calthorpe Broad and Martham North and South.
Calthorpe Broad and Hickling Broad:	National Nature Reserve (NNR)

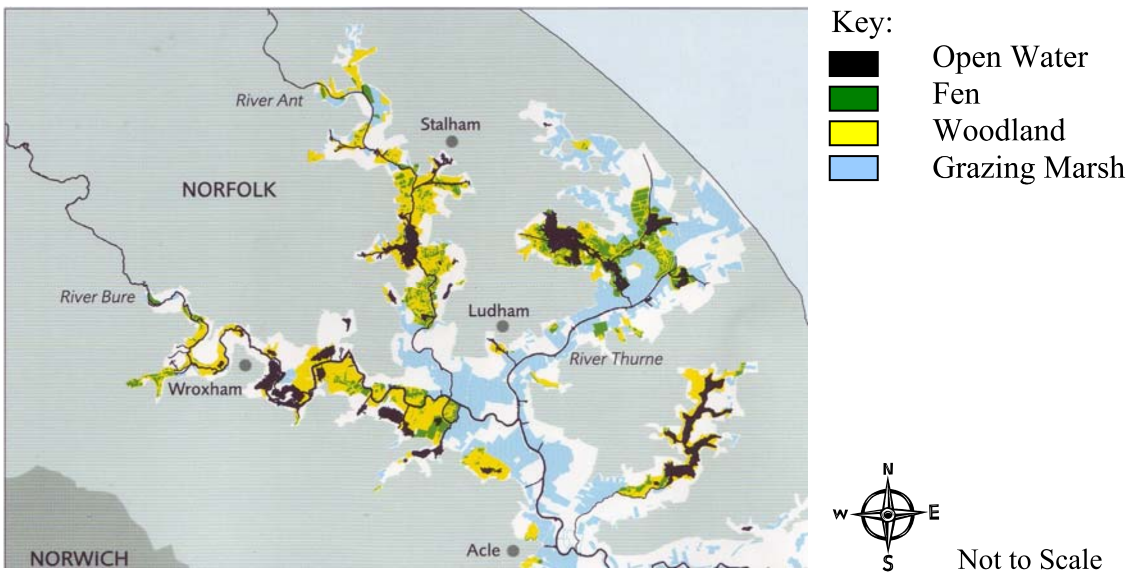


Figure 3.1.5: Habitats within the Broads

There are several County Wildlife sites within the Upper Thurne area. All of the above fall within the Broads Special Protection Area (SPA), candidate for Special Area of Conservation (cSAC) and the RAMSAR wildfowl designations.

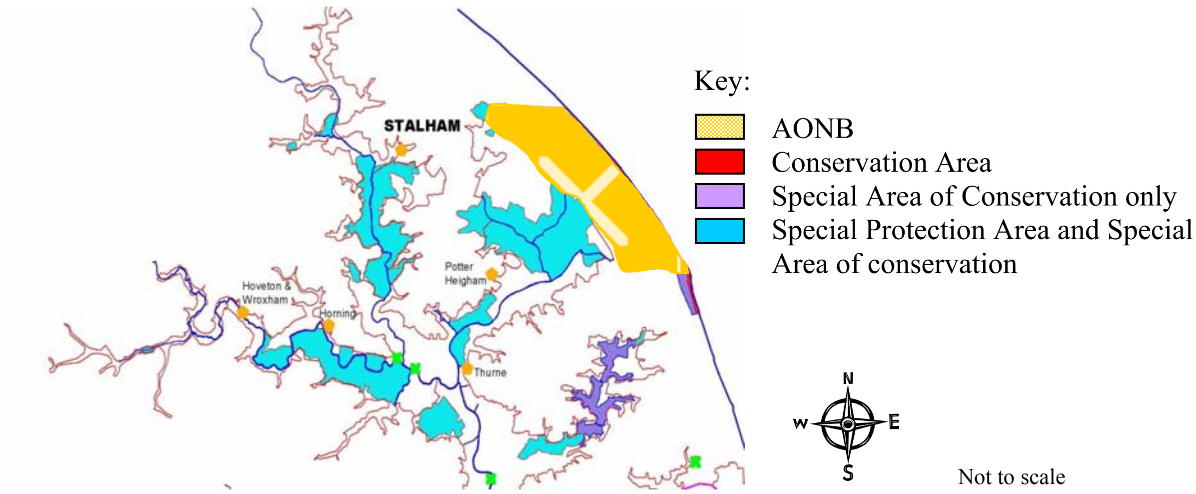


Figure 3.1.6: Designated protected areas within the Broads

3.2 Hydrology

3.2.1 Surface Water

Rivers

The river basins of the Broadland rivers are large for the UK (Table 3.2.1) with the Bure, Yare, Wensum and Waveney together draining a substantial part of East Anglia. The rivers have very gentle gradients as they flow through Broadland, approximately 3cm per kilometre (George, 1992). However, their water regime is complicated by tidal influence and also by differences in run-off rates (Table 3.2.2). These are due to variations in the permeability of the soil types which predominate in the river basins; those of the Bure being relatively permeable to rainfall, with an estimated annual infiltration rate of 143mm (East Suffolk and Norfolk River Authority, 1971). The mean discharge rates of the rivers are given in Table 3.2.3.

Table 3.2.1: River basins areas (East Suffolk and Norfolk River Authority, 1971)

River basin	Sub-catchment	Gross Area (km^2)
River Bure	River Bure	330.7
	Spixworth Beck	61.5
	North Walsham & Dilham Canal	49.3
	Tidal River Bure & River Ant	164.2
	Tidal River Bure & River Thurne	271.7
	Total	**877.4**

Table 3.2.2: Rates of river basin run-off (Anglian Water Authority, 1981)

River	Bure	Ant
Gauging Station	Horstead Mill	Honing Lock
River basin Area (km^2)	313.0	49.3
Period of Observation	1974-1982	1971-1981
Mean run-off (mm)	219.0	191.1
Maximum run-off (mm) & year	272.4 - 1981	221.5 - 1981
Minimum run-off (mm) & year	164.5 - 1974	157.3 - 1973

Table 3.2.3: Mean discharge rates (calculated from EA daily flow data 1978 – 2000)

River & Location	Measured (M) or Estimated (E)	Mean Discharge ($m^3 s^{-1}$)
River Bure – Horstead Mill	M	2.38
River Bure – confluence with River Ant	E	3.2
River Bure – confluence with River Yare	E	6.2
River Ant – Honing Lock	M	0.32
River Ant – confluence with River Bure	E	1.1
River Thurne – confluence with River Bure	E	0.96

The River Bure rises at Melton Constable at a height of 80m OD and flows east to Blickling Mill then south to Horstead at its tidal limit, finally flowing east once more to the sea at Great Yarmouth (Fig 3.2.1). There are a number of large tributaries above Wroxham; Scarrow Beck flowing south to join the Bure below Blicking Mill, Kings Beck flowing south and joined by its tributary Stake Beck before its confluence with the Bure above Buxton Mill, and Spixworth Beck which flows east to join the Bure above Wroxham. River flow is continuously monitored at two gauging stations at Ingworth and Horstead Mill.

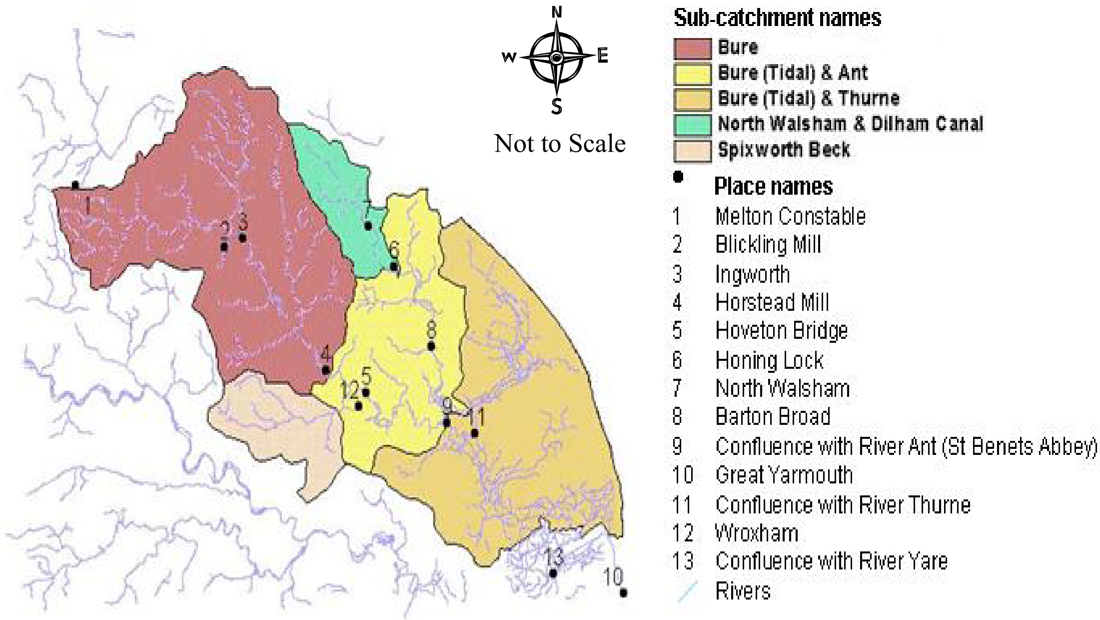


Figure 3.2.1: Map of the River Bure sub-basins

In part of the Bure's course its long term discharge is only approximately 3 $m^3 s^{-1}$, with a water retention time of about 5.4 days (Moss *et al.*, 1989). Winter high flows are on average less than double the summer flows. Discharge at Ingworth in January –

March averages 4.3 $m^3 s^{-1}$ and in July – September 2.4 $m^3 s^{-1}$. The contribution of the River Ant increases the Bure flow by about one-third; there is little addition of new water below Hoveton Bridge. There is almost certainly some ground water seepage, but in the summer when evaporation greatly exceeds precipitation in the river basin, this is very small.

The River Ant flows southwards through Norfolk. It rises near Honing at approximately 59.0m OD and eventually joins the River Bure west of St Benets Abbey, having passed through Barton Broad. In one stretch it was straightened and deepened to form a channel in the late nineteenth century, with locks at North Walsham and Honing. The Ant is a very narrow winding river, approximately 1.5m deep (Gurney, 1911) with an average discharge of 0.32 $m^3 s^{-1}$.

The River Thurne nominally drains an area of about 109km^2 and for several centuries it has not taken the shortest route to the sea, instead flowing inland towards the River Bure. The River Thurne receives gravitational drainage from only small areas of the river basin since drainage of the marshland, and associated peat shrinkage, have left the river standing above the general ground level. This results in the river having a very low natural discharge which is compounded by its very low gradient (0.002%) (Holman & Hiscock, 1998). The River Thurne has several sources (Fig 3.2.2):

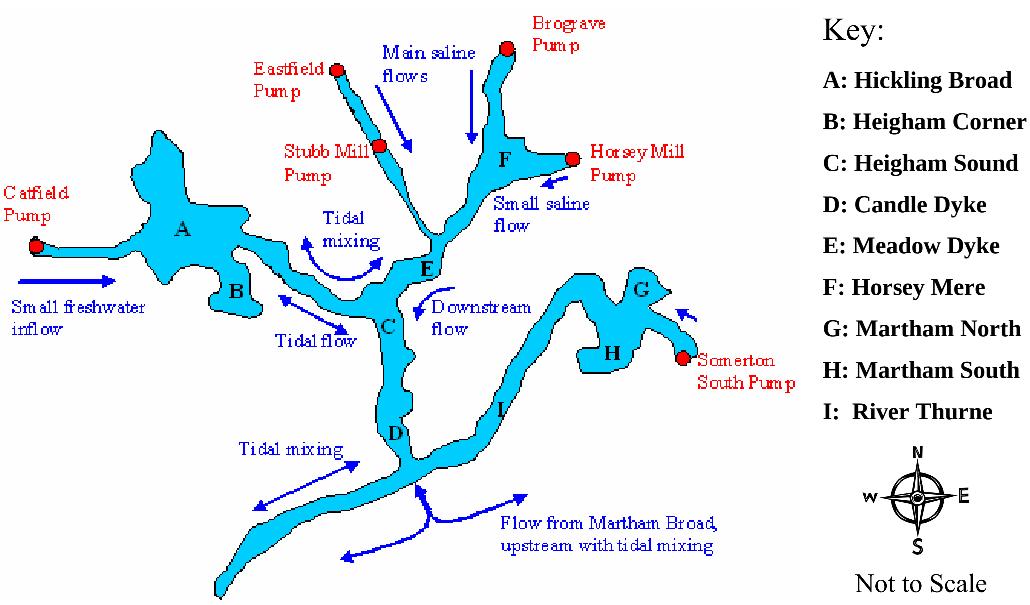


Figure 3.2.2: Diagram of the Upper Thurne System (Bales *et al.*, 1993)

- One lies near East Somerton and feeds Martham Broad
- The main sources of water for the Thurne system discharge into Horsey Mere and Meadow dyke, draining an area north of Hickling Broad and Horsey Mere bordered by the coast.
- A minor source drains land near the village of Catfield and discharges through Catfield dyke into Hickling Broad.
- The outflows from Horsey Mere and Hickling Broad converge in the riverine Broad – Heigham Sound.
- Heigham Sound has its outlet in Candle Dyke, which joins the River Thurne 7km above its confluence with the River Bure.
- Between these points is a smaller Broad, Womack Water, which also discharges into the River Thurne.

Broads

There are a total of 39 broads in the study area. Apart from the headwater sites such as Hickling Broad and Horsey Mere the broads fall into two categories. 'Side-Valley' sites which are situated in tributary valleys and 'by-passed' broads, which are located to the side of the main river (Gregory, 1892).

The hydrology of the side-valley broads is fairly simple, the majority being fed by small tributary streams. All the side-valley sites and some of the by-passed broads are separated from the main river system by sluices. These are installed to maintain a reasonable depth of water for supply or angling purposes, without interfering with the drainage of the adjoining marshland.

The water regime of the broads which are in open communication with the rivers is much more variable and complex. In some cases, the channel or channels connecting the site with the river are so wide that the water in them is replaced very frequently, especially if the system is subject to strong tidal action. In contrast, sites such as Snape's Water, which are connected to the main river system by narrow, often heavily silted dykes have much longer retention times (George, 1992).

Moss *et al.* (1984) showed that relatively rapid mixing of water masses takes place in sites such as Wroxham Broad, and that wind-induced currents, as well as tidal action are responsible for this. Results showed that Wroxham Broad has a retention time of about 4 weeks during dry spells in the summer, but this decreases to 2 weeks in wet summers, and to 1 week in the winter. These figures represent mean flushing rates of about 4, 8 and 12 per cent a day respectively. In contrast further results showed that Hoveton Great Broad has a theoretical replacement time of between 6 and 8 weeks; however, retention time will be much greater during dry periods in the summer.

The Upper Thurne system is more hydrologically complex as can be seen in Figure 3.2.2. Water enters Horsey Mere and Meadow Dyke through the land drainage pumps, mostly in winter; little new water enters in summer when evaporation rates are high. Not much water enters Hickling Broad directly at any time; what does, comes through Catfield pump, which serves only a small part of the river basin area. The winter water is pushed into Hickling Broad by the flood tides and replaces progressively that left from the previous summer; most of the water pumped from the river basin does not enter Hickling Broad.

In summer, water may be moved between Hickling Broad and Heigham Sound and Horsey Mere by tides, but little new water is likely to enter the system. The tidal range at Hickling Broad is about 4cm, equivalent to 3-4% of its volume. Exceptionally high tides are capable of exchanging 25% of the volumes of Hickling Broad and Horsey Mere within 2-3 days (Holdway *et al*., 1978).

The winter high water drops gradually from February and then rapidly in May to a low level through July. The levels then pick up again steadily to the winter level in November, as can be seen in the graph below. There was a general increase in water levels in the period from 1993 - 2000 (Hoare, 2002).

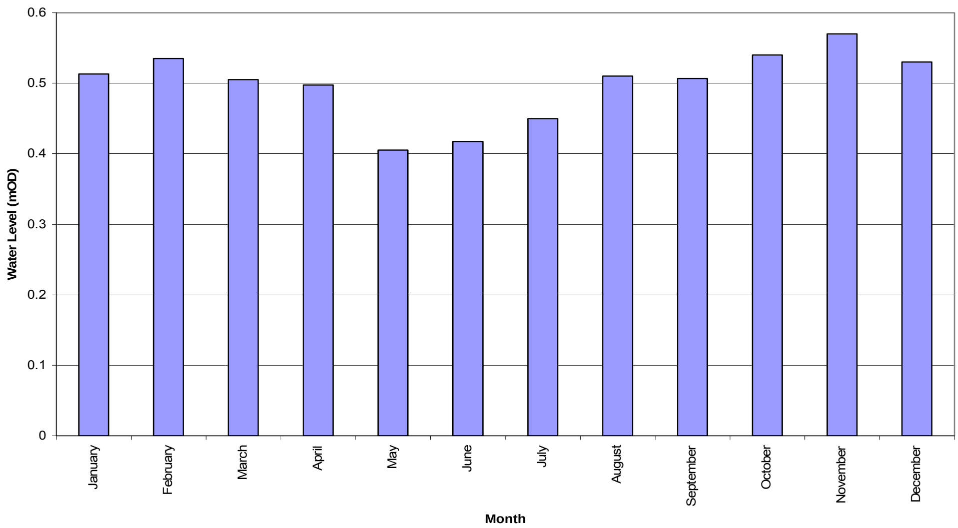


Figure 3.2.3: Monthly mean water level from Hickling Broad (1993 – 2000)

3.2.2 Land Drainage System

Land drainage of the marshlands enables the Thurne river basin to be one of the most intensively farmed areas in the Broads (Watson, 1981). In the marshes where cattle grazing is practised, a high water level is required to provide water for the cattle and to maintain a water table for grass. Conversely, to achieve high arable yields a freeboard of about 1.25m is required throughout the year, together with levelling and under drainage of the marshes (Holman & Hiscock, 1998).

The Internal Drainage Boards (IDB) controls Land drainage. Two IDBs operate pumps in the Upper Thurne area; the Smallburgh IDB, with Catfield, Stubb Mill and Eastfield pumps; and the Happisburgh to Winterton IDB with the Brograve, Horsey Mill, Somerton North and Somerton South pumps. They are responsible for draining all land below 2.74m and therefore the combined drainage area of the pumps totals approximately 4180 ha (Hoare, 2002).

A bank to prevent the flooding of neighbouring areas if the dunes or river were to be breached encloses each drainage area. The water from the large internal networks of surface drains and tile drains flows into the arterial main drains where 13 drainage pumps electrically pump it into the external system of connected rivers and lakes.

Each pump has an attendant who is responsible for maintenance and recording information from the pump on a regular basis. This information includes the electric meter readings, water level on the drain side and hours pumped (where available). These data have also been obtained from the Environment Agency, although records were not complete for all pumps and the Agency do not hold data on the rest of the pumps in the Thurne river basin.

Table 3.2.4 shows the annual water balance for the Thurne river basin. It demonstrates that the land drainage system forms a highly significant component of the river basin hydrology. The drainage pumps discharge the equivalent of the hydraulically effective rainfall falling on the entire Thurne river basin. It is evident that land drainage is also, by at least an order of magnitude the largest groundwater abstractor in the Thurne river basin (Holman *et al*., 1999).

Table 3.2.4: Water balance calculations for the River Thurne river basin (Holman *et al*., 1999)

	Year 1 (1991/1992) Volume (x 10^3 m^3)	**Year 2 (1992/1993) Volume (x 10^3 m^3)**
Inflows		
Rainfall	59896	72409
Saline intrusion	658	957
River leakage	918	1018
Mains water leakage	165	165
Irrigation returns	0	0
Total inflows	61637	74549
Outflows		
Evapotranspiration	46201	57610
Open water transpiration	334	335
Drainage discharge:		
Freshwater	9920	16454
Saline water	658	957
River leakage	918	1018
Groundwater outflow	55	51
Groundwater abstraction	215	91
Total outflows	58301	76516
Storage changes		
Soil moisture	1515	-1908
Groundwater	371	186
Surface water	16	-15
Total storage change	1902	-1737
Total discrepancy	1434	-230

3.2.3 Tides

Although the emphasis of this study is the Upper Thurne river basin the Bure and Ant river basins are also being considered because of the complex hydrological connection between the waters of the Bure, Ant and Thurne due to the twice-daily tidal incursion.

Recordings early in the 20th century describe tidal ranges of 23 – 37.6cm at Acle Bridge, 19cm at Thurne Mouth, 17.6cm at St Benet's Abbey, 9.7cm at Horning Ferry and 2.5 – 8.4cm near Hoverton Great Broad (Gurney, 1911). Current tidal ranges are similar to those recorded by Gurney. At Great Yarmouth the range is approximately 1m in summer and 2.7m in autumn and winter, rising higher in periods with northwesterly gales. At Acle Bridge it is normally 26cm, exceptionally 45cm and at Hudson Bay it is 8 – 12cm, rather less than the 20 – 30 cm changes which frequently occur as a result of changes in river flow (Moss *et al*., 1989).

On the River Thurne the tidal range at Womack Water, 1.25 miles from the Thurne mouth is 16.61-19.81cm and at Potter Heigham Bridge, which is 3 miles from the mouth, it is approximately 12.7cm (Gurney, 1911). At Heigham Sound a tidal gauge was used three times by Gurney (1911) to give an average range of 4cm, with averages for the three sets of charts of 2.54, 3.81 and 5.5cm. Therefore these charts show great irregularity in the tidal range at Heigham Sound. Further upstream at Deep Dyke the current varies in direction, sometimes flowing out of Hickling Broad and sometimes into the Broad, but it has been shown that this flow does not correspond with the tide (Watson, 1981).

The tidal range increases progressively down the whole system and there are considerable cyclical level movements and mixing in the lower reaches and at the confluence of the River Bure and Thurne. These movements between the Bure and Thurne have been traced by dye additions (Rhodamine WT) (Moss *et al*., 1989). They indicated that sewage effluent discharges from points far downstream could influence water quality several kilometres upstream. Moss *et al.* (1989) placed 2.4 kg of Rhodamine WT in the River Thurne at the start of the ebb tide on 22nd October 1980. The trace first moved into the River Bure below Thurne mouth and then back into the Thurne during the flood tide later that day. No dye entered the Bure above the Thurne mouth. On the next flood tide, which was slightly higher, approximately 40% of the dye returned to the Thurne. This went as far upstream as Ludham water and 60% entered the Bure above the Thurne mouth and penetrated for approximately 1.5km.

The implication of this experiment is that water may be retained in this part of the system for weeks before it moves out of this stretch to the lowest part of the system.

In this period it may move between the rivers Thurne, Bure and Ant several times. Long retention times in the main channel have also been confirmed by Moss *et al.* (1989) by following rates of chloride dilution after high tides, which gave values for retention time of 3-4 weeks.

It is clear from the above research that water from the Rivers Bure and Ant regularly enters the River Thurne, travelling as far upstream as Potter Heigham. Therefore nutrient loads from the Bure and Ant river basins are also transported into the Thurne river basin although the extent of this is unknown due to the possible restriction of flow by the bridge at Potter Heigham. The effect of this hydrological connection needs to be considered and investigated further (Parallel PhD - Sofía Martínez).

Chapter Four Water Quality in the Hickling System

4.0 Nutrients

The ecology of the Broads is heavily dependent on the chemistry of the water reaching it. The two main nutrients in the study area are phosphorus and nitrogen. Nitrogen is a major plant nutrient and is often applied in large amounts to agricultural land to maintain optimal yields. To ensure that plant nitrogen availability does not limit crop yields additional nitrogen is generally added to agricultural crops as inorganic fertilisers or in organic forms such as farmyard manure, slurry or sewage sludge. When nitrogen fertiliser is applied in excess of plant requirements it undergoes a series of transformations and transfers in soil, which can lead to pollution of the waterways and gaseous emissions from soil. Nitrogen may also be introduced into the system by nitrogen fixing plants, rainfall and directly from the atmosphere in the form of ammonia, nitrogen oxides and nitrate.

Phosphorus is one of the most important mineral nutrients for biological systems, yet it is also one of the least available nutrients in terms of its demand in terrestrial and aquatic environments (Jarvie *et al.*, 2002). Therefore, mineral phosphate fertilisers and animal manures are applied to agricultural land to raise soil phosphorus levels and maintain crop yields. As well as commercial fertilisers and animal manures phosphorus may be introduced into the environment by plant residues, rainfall, municipal agricultural and industrial wastes or by-products in addition to the natural weathering processes of soil minerals.

4.0.1 Sources and Movements

Point sources of nutrients such as effluent from STW (Sewage Treatment Works), industrial process effluents and discharges from septic tanks are not particularly significant in the mainly agricultural catchment of the Upper Thurne; however they play a more prominent role in the Bure catchment.

There are no STW or process industries which discharge to the Thurne catchment. However there are scattered septic tanks and small sewage digesters with EA

(Environment Agency) consented discharge licences. These are small inputs over a wide area and occur at low density so their impacts on nutrient levels in the system as a whole represent a more diffuse pollution source. There are many small sewage treatment works in the Bure catchment which do not undertake phosphate removal though effluent discharged from the larger works (Aylsham, Belaugh, Briston and Rackheath) all undergoes phosphate removal. Discharges from these works with phosphate removal have a quality target of an annual average of 1mg l^{-1} in the final effluent (Madgwick, 1999).

The River Bure can be considered in two parts, separated at Swan Bend, above which the tidal influence is small. In the lower part of the catchment there are two further sewage treatment works, Acle and Caister. Neither of these have phosphate removal as it is deemed that these works 'do not require treatment for improvement of water quality' (Broads Authority, 1982).

Significant phosphorus loading was however added to the Thurne system in the late 1970's, which the above identified sources could not account for. This was attributed to guano from a large roost of black headed gulls and consequent phosphorus release from sediment (Hoare, 2002). The levels of P in the sediment have gradually been declining since the refuse tip, which the gulls visited in the day, was closed. The remaining major source of diffuse nutrients is run-off from agricultural land e.g. manures and chemical and green fertilisers. Nutrient losses from these can also be affected by on farm management.

The Bure valley catchment is intensively farmed; therefore non-point sources of nutrients are important. Johnes (1996) looked at the nutrient inputs and transport dynamics in the Bure catchment at eight sample sites (Fig 4.0.1). All the sites showed seasonal patterns in nitrogen concentrations suggesting that nitrogen reaches the river through predominantly non point sources except for at Bickling Mill. Data from this site showed no seasonal pattern for nitrogen or phosphorus concentration. The majority of the sample locations showed no seasonal phosphorus patterns, this was however only over a one year period (1995-1996). This signified that the main source of phosphorus concentrations in the catchment was from point sources such as septic tanks and smaller STW's. Spixworth Beck had the lowest phosphorus concentration in the catchment reflecting the predominantly non point source of total phosphorus in this tributary.

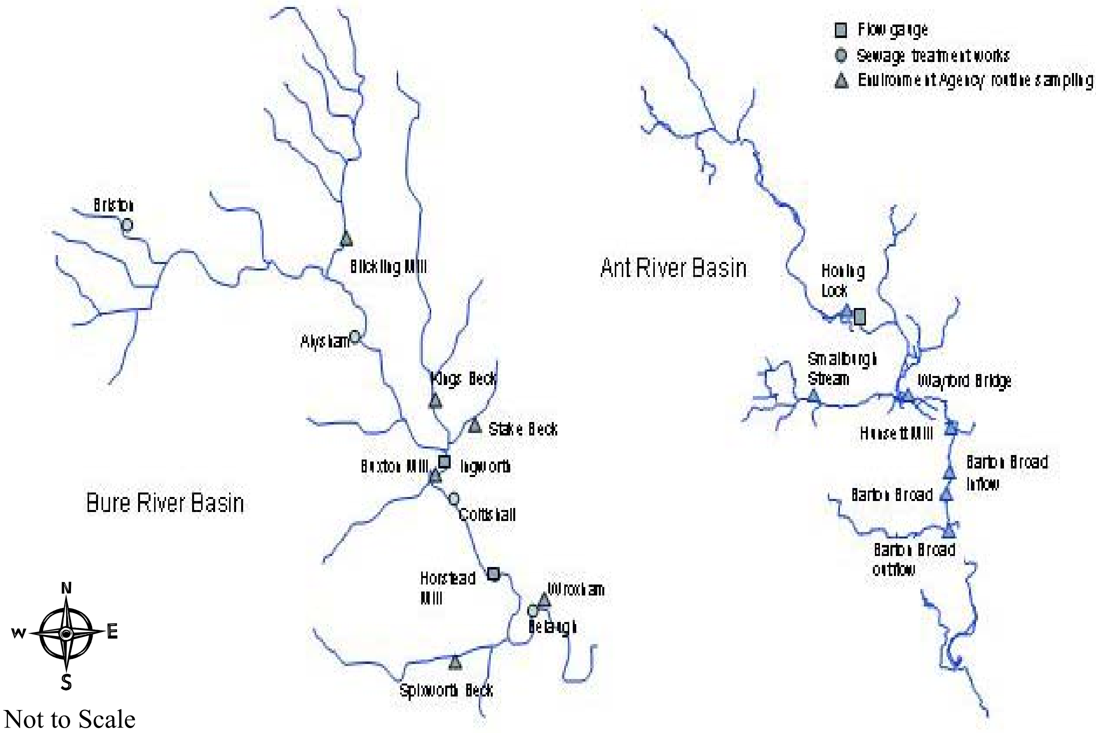


Figure 4.0.1: EA sample sites, gauging stations and STW's in the Bure and Ant river basins

All four major treatment works in the Ant catchment (North Walsham, Stalham, Worstead and Horning) along with all industrial effluent sources undergo phosphorus removal before discharge into the river (UK Water Resources Act, 1989). Flow from the North Walsham sewage treatment works has been redirected to a coastal outfall at Mundesley in 1980.

Work carried out by Phillips *et al.* (1999) suggests that the installation of tertiary chemical dosing at Stalham in 1977 to remove phosphorus from the final effluent has had a very small impact on the river load, as the initial impact of the Stalham discharge on Barton Broad was relatively small (Osborne, 1981) . In comparison the diversion of the North Walsham sewage treatment effluent in 1980 resulted in a substantial 90% reduction in the discharged phosphorus load. Total river phosphorus load clearly responded to this change, although the 90% reduction in discharge load was only matched by a 50% reduction in river load at the point where it enters Barton Broad (Hunset Mill). Osborne (1981), Moss *et al.* (1988) and Johnes (1996) all concluded that the discrepancy between discharge load and reduction in river load was due to the uptake of phosphorus by sediment. This sediment can move downstream under flood conditions, undetected by spot-sampling regimes, ultimately to be deposited in Barton Broad where subsequent chemical and biological transformation can make it available to the overlying water (Phillips *et al*., 1994).

Since 1980 further upstream at Honing Lock, the river load has been higher than that discharged from remaining upstream point sources, revealing the presence of other non-point source inputs. Further downstream the discrepancy between river load and upstream point sources are less apparent.

Phosphorus release from sediments is of fundamental importance in relation to the nutrient enrichment problem on the Broads. Phosphorus release occurs during much of the year and varies considerably in different Broads and river sediments. The rate of the release is controlled by two principal factors: the phosphorus concentration in pore water and the amount of ferrous iron in the sediment (George, 1992).

Considerable research has been carried out into nutrient export in the Broads. These have in the past considered phosphorus as the controlling nutrient causing eutrophication in the Broads. This is because in most fresh water systems phosphorus is the limiting nutrient. Therefore an increase in phosphorus loads can affect the composition and diversity of the aquatic ecosystem. Consequent attempts to restore shallow lakes have seen increasing plant populations, however these are usually very limited in species and the restored sites frequently revert to algal dominance after a few years (Meijer *et al*., 1999). One possible reason for this is that high nitrogen levels lead to low plant diversity. Such low diversity communities are vulnerable to damage early in the season through natural variations in weather or pressures from grazing waterfowl (James *et al*., 2003). Restoration of stable diverse plant communities might then require nitrogen as well as phosphorus control. This factor needs to be considered when looking at future land and water management scenarios.

In terms of nutrient movement the low rates of rainfall in Norfolk mean that soluble reactive phosphorus export along through-flow pathways will be limited. However, export along field drains may be a significant pathway during wet periods, and particularly during snow melts, when soluble phosphorus in excess of adsorption capacity may be flushed along macropore pathways into field drains and into adjacent watercourses. True overland flow is unlikely in the Norfolk region, given the low degree of slope, and low annual rates of rainfall. Some overland flow may be generated in the stream corridor on arable land, particularly where direct grazing of fodder crops by livestock takes place, since this will compact surface soil horizons, reducing infiltration capacity and porosity, but it is unlikely to be the dominant pathway. However, much of the region is under drained to allow arable cultivation to take place on low-lying land. When combined with the high proportion of fine particulate matter in the loamy soils of this region, selective transport of phosphorus adsorbed to fine particulate matter is likely to be an important pathway for the transport of phosphorus to Norfolk Rivers. Therefore the transport of these nutrients to watercourses can take one of several routes in the study area:

- Significant amounts are transported in tile drain flow.
- Other processes which mobilise nutrients are poaching of dyke margins by livestock, dyke clearance and deepening work, excessive manure applications during wet or frozen conditions
- The route which nutrients take from the agricultural catchment of the Upper Thurne itself is via the drainage network and finally through the IDB pumping stations (Hoare, 2002).

4.0.2 Dynamics

In order to calibrate a catchment scale model of the study area a detailed record of total N and total P loading is required. In addition an understanding of the nutrient transport dynamics of the rivers will provide a valuable insight into the likely origins of nutrient loading. If little seasonal pattern were to emerge for P fractions, for example, this would suggest that the origins of P loading were from non-seasonal point source discharges rather than from non-point catchment sources. This pattern will also be influenced by in stream transformations of N species and P fractions, but to lesser extent than in standing waters, since river hydrochemistry is largely determined by flow and sediment transport in all but the slowest flowing rivers.

In all three catchments river water quality data from the Environment Agency have been obtained for the period 1981 – 2000. Figure 4.0.2 shows the location of these sites and Table 4.0.2 describes the dynamics of these sites. Data from 1998 are shown graphically in Appendix One to aid the understanding of the nutrient dynamics in the system.

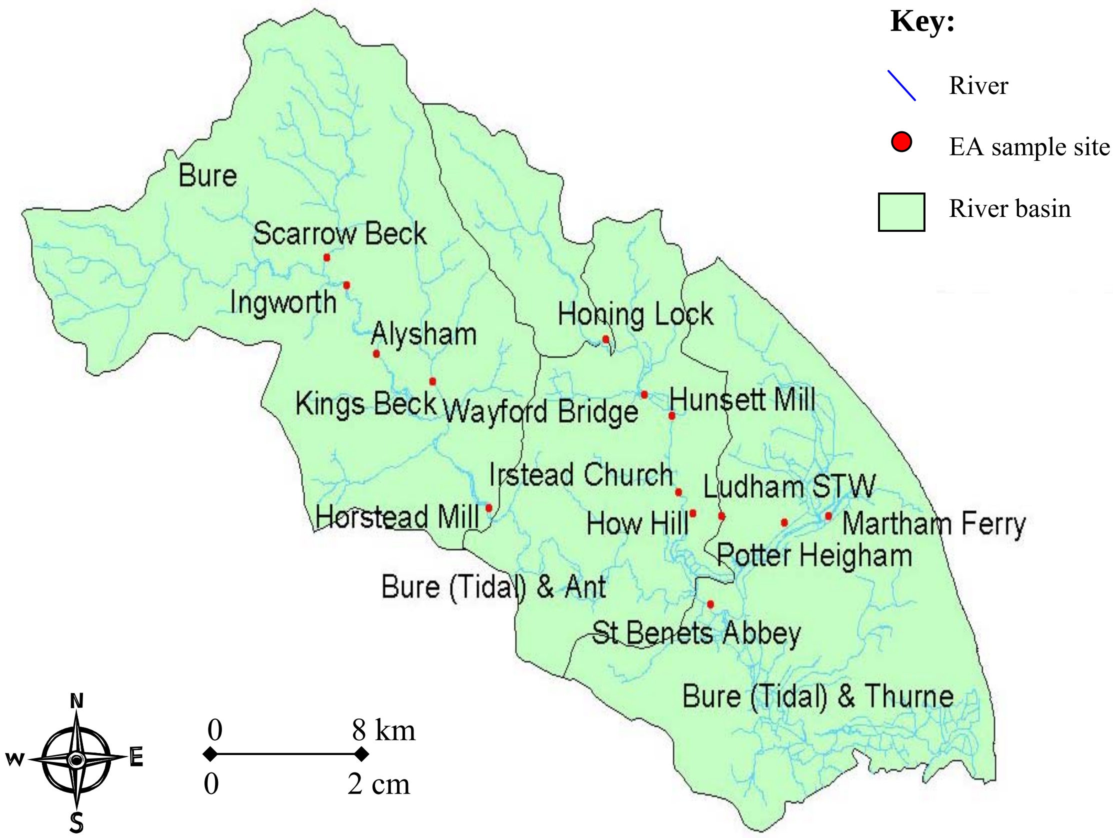


Figure 4.0.2: Location of Environment Agency sample sites

From Table 4.0.2 it can be seen that despite tidal incursion, water quality data for oxidised N in the Bure catchment shows a seasonal pattern with winter maxima associated with periods of high catchment run-off, and summer minima reflecting lower rates of run-off, plant uptake and export to the atmosphere through denitrification in the slower reaches. The pattern is not extreme, reflecting the subdued river regime of the rivers in the Norfolk regions. Ammonium concentrations are highest during the spring period, reflecting spring storm flow, combined with periods of low flow and low oxygen saturation in the river. Overall the N species show a weak seasonal pattern with a winter maximum, suggesting that N export to the Bure is largely derived from non-point sources in the catchment, with a small percent of the total load derived from point source discharges from STWs, notably at Belaugh.

Overall the P data for the Bure show no strong seasonality and suggest that point source discharges are likely to be the dominant source of P loading on the River Bure. However EA routine sampling only takes place bi-monthly therefore the P load from event-based flow in which the majority of P load would be exported from non-point sources was unlikely to be recorded at this sampling interval.

The Ant is a small river with a catchment, which is less intensively farmed than that of the Bure. This is reflected in much lower nitrogen concentrations than those observed in the Bure. P concentrations are however similar to those seen on the Bure, but a more defined seasonal trend can be observed, suggesting that P concentrations are dominated by non-point source origins.

The Thurne catchment has no STW's discharging above Potter Heigham Bridge. N species show seasonal trends of winter maxima and summer minima. In contrast to the Bure and Ant catchments ammonia also shows a seasonal trend of summer minima and winter maxima, this is due to the influence of peat drainage in the catchment. Phosphorus shows no real seasonal trend but does however have the lowest concentrations in all three catchments. The nitrogen levels are modest compared to the other two catchments, as the Thurne catchment is not so intensively farmed. Most nutrients that enter from the catchment do so in winter when pumped volumes are the highest.

Table 4.0.2: Nutrient dynamics in the study area (adapted from Johnes, 1996b)

Site	Nitrogen Trends	Phosphorus Trends	Reason
River Bure at Scarrow Beck	N-oxidised & N-inorganic show clear seasonal trends with winter maxima and summer minima	No clear trends	Overall no clear trends likely caused by a greater contribution of nutrients from human sources in the catchment
River Bure at Kings Beck	Nitrogen speciation show marked seasonality, with winter maxima and summer minima	No clear trends	Clear nitrogen speciation trends and very few human settlements suggest the importance of non-point sources in this catchment
River Bure at Aylsham	Ammonia repeats summer maximum pattern, mostly recorded as < 0.5 mg l^{-1}. Increase in oxidised N (highest for all sample sites)	No clear trends	High oxidised may reflect different balance in arable and livestock production in this part of the Bure catchment. Ammonia recorded as < 0.5 mg l^{-1} reflects higher oxygen saturation in the faster flowing waters at this site. Few human settlements so trends may reflect a greater proportion of livestock waste utilised as organic fertiliser
River Bure at Horstead Mill	Weak seasonal pattern	Weak seasonal pattern	Weak seasonal trends reflect export from non-point sources. Horstead lies upstream from 3 major STW's
River Bure at Wroxham Rail Bridge	Weak seasonal pattern	No strong seasonality	Weak seasonal N species patterns suggest that N export is largely derived from non-point sources. No strong P trend suggests point sources discharges are likely to be the dominant source
River Ant at Honing Lock	Lower N concentrations then those in the Bure	P concentrations similar to the Bure, but more defined seasonal trends	Less intensively farmed catchment reflects in lower N concentrations. Seasonal P trends suggest P concentrations are dominated by non-point sources

Site	Nitrogen Trends	Phosphorus Trends	Reason
River Ant at Wayford Bridge	Nitrogen has a weak seasonal trend. Ammonia trend is out of line with trends observed in other catchments	Weak seasonal trend	Lower concentrations for both P and N are a result of lower proportion of major urban areas and lower proportion of intensive livestock production
River Ant at Hunsett Mill	Sparse data	No seasonal pattern	No seasonal P pattern suggests that the majority of P comes from point sources. This site lies downstream of STW at Honing Lock, although phosphorus stripping takes place at this works
River Ant at Irstead Church	Ammonia shows no real trend. Oxidised N and inorganic N show winter maxima and summer minima	No data	Winter N maxima trends are associated with periods of high catchment run-off
River Ant at How Hill	No data	Summer maxima	Non-point sources are important in this catchment
River Thurne at Martham Ferry	N species show winter maxima and summer minima. Unlike the Bure and Ant ammonia also shows seasonal trends	No seasonal trend. Lowest concentrations in study area	Ammonia seasonal trends are due to the influence of peat drainage. Lower P and N concentrations are a result of a less intensively farmed catchment. Most nutrients enter the catchment in winter when pumped volumes are the highest
River Thurne at Potter Heigham	Weak N seasonal trend. Ammonia similar to Martham Ferry	No seasonal trend	Weak seasonal trends suggest N export is largely derived from non-point sources
River Thurne at Ludham STW	High N concentrations with no seasonal trend. Ammonia is recorded at < 0.5 mg l^{-1} for most of the time	No seasonal trend	Ludham STW discharges upstream of this site reflecting the lack of seasonal trends. Ammonia recorded as < 0.5 mg l^{-1} reflects a higher oxygen saturation in the faster flowing waters at this site due to discharge from STW

4.0.3 Changes

Levels of both nitrogen and phosphorus have changed in both the Ant and Bure catchments over time (Fig 4.0.3 and 4.0.4).

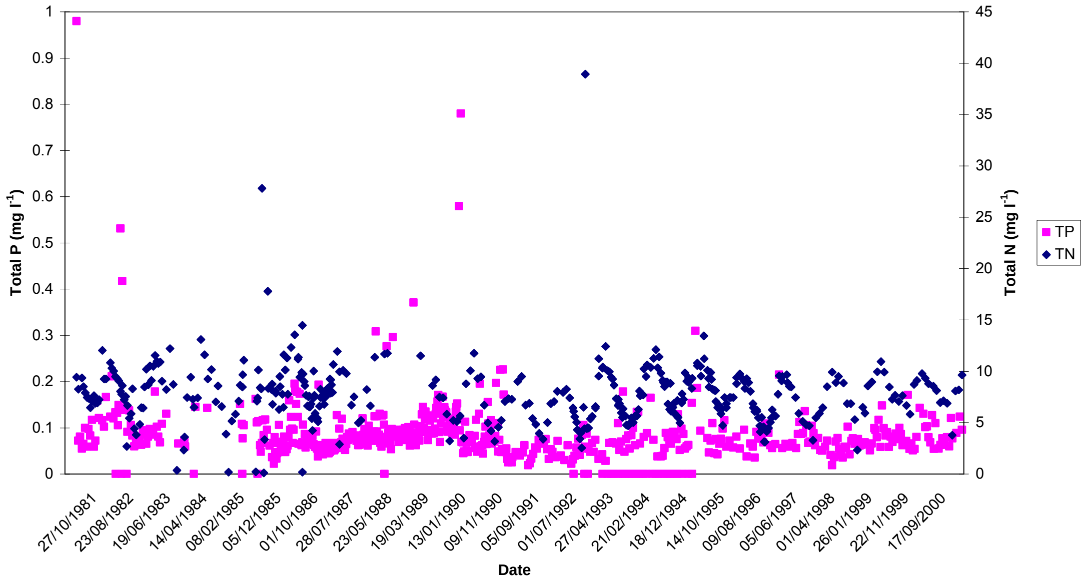


Figure 4.0.3: Nutrient concentrations at Honing Lock, River Ant (1980 – 2000)

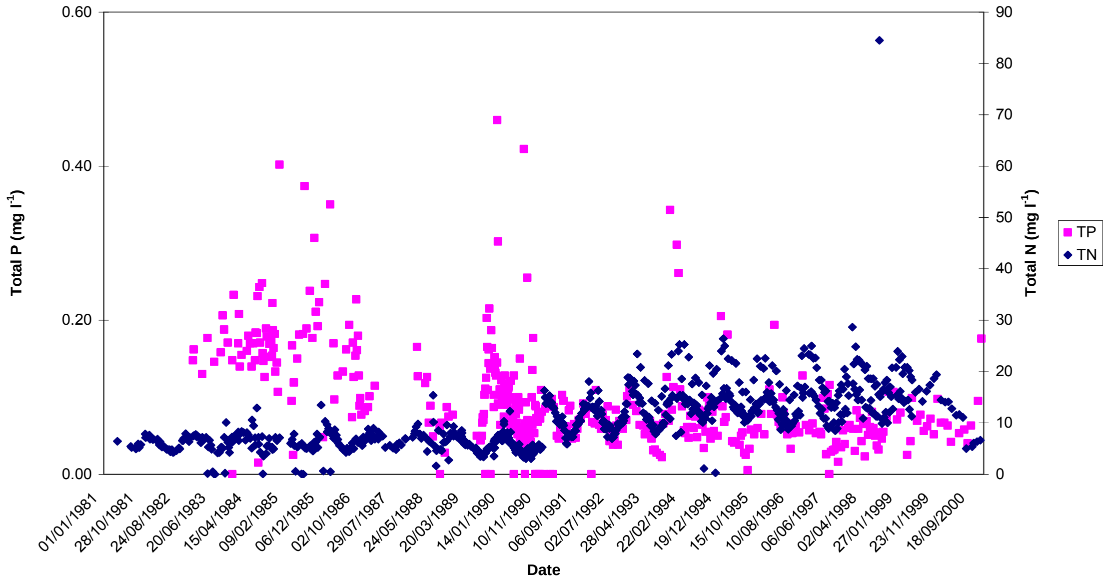


Figure 4.0.4: Nutrient Concentrations at Horstead Mill, River Bure (1980 – 2000)

Although the graphs in Figures 4.0.3 and 4.0.4 suggest there has been a change in nutrient concentrations statistical analysis using the Student T test showed overall there has been no significant change in nutrient concentrations (Table 4.0.3). At Horstead Mill on the River Bure there has however been a significant increase in nitrogen levels between 1990-2000 and 1981-2000, there has also been a significant overall increase in phosphorus levels between 1981-2000. The use of the Student T test on the River Ant data for the same periods showed no significant change in nitrogen or phosphorus concentrations.

At both Horstead Mill and Honing Lock seasonal patterns can be seen in the nutrient concentrations. At Horstead Mill from 1993 onwards a lower and upper trend in nitrogen can be observed. This may be a result of the increased sampling frequency at this site therefore capturing more event based nitrogen movement.

Table 4.03: Statistical analysis on long term nutrient data for the River Bure (Horstead Mill) and Ant (Honing Lock)

			Analysis Dates		
Site	Parameter	Statistic	1981-1990	1990-2000	1981-2000
Horstead Mill	N	Degrees of freedom	85	86	21
		T_{cal}	0.44	6.49	3.36
		T_{tab}	1.98	1.98	2.08
		Significance at 5% level	No	Yes	Yes
	P	Degrees of freedom	83	82	19
		T_{cal}	0.05	0.42	6.08
		T_{tab}	1.98	1.98	2.09
		Significance at 5% level	No	No	Yes
Honing Lock	N	Degrees of freedom	27	23	26
		T_{cal}	-1.52	-1.02	0.42
		T_{tab}	2.05	2.06	2.05
		Significance at 5% level	No	No	No
	P	Degrees of freedom	63	66	35
		T_{cal}	-1.86	0.58	-1.12
		T_{tab}	2.00	2.00	2.04
		Significance at 5% level	No	No	No

4.1 Water Quality and Data Analysis

An initial analysis of flow and water quality contaminants (Total Phosphorus and Total Oxidised Nitrogen) has been undertaken to help recognize simple temporal and spatial varying patterns in nutrient concentrations. The aim of this was to gain a better understanding of the system to aid the modelling of the system and to consider the implications of the system for modelling.

4.1.1 Initial Flow Analysis

As discussed previously the study area is very complex in terms of hydrology, especially the Thurne River basin. An attempt has however been made to understand the flow patterns within the system. Rainfall in the study area is low in the UK context, averaging 520.6 mm yr^{-1}. Evapotranspiration is, by contrast, high in the UK context at an average 455 mm yr^{-1}. As a result hydrologically effective rainfall, as a key driver for the generation of flow within the river channel and nutrient and sediment transport from land to water, is relatively low (as mentioned in Chapter Three) at 142 mm. This produces a mean annual flow of 0.30 m^3 s^{-1} at Honing Lock and 2.38 m^3 s^{-1} at Horstead Mill, both of which are low when compared with the rate of flow generated in other UK river basins of similar drainage area.

Figure 4.1.1 and 4.1.2 show flow duration curves for the Rivers Ant and Bure. It can be seen that the curve for the River Bure slopes steeply which suggests that the river has highly variable larger flows. In contrast the River Ant has a gently sloping curve, which indicates a large delayed flow component. Both curves flatten out considerably at higher percentile flows. This may represent the perennial storage in the two drainage basins, with the River Ant showing a longer and flatter curve and hence the larger amount of storage.

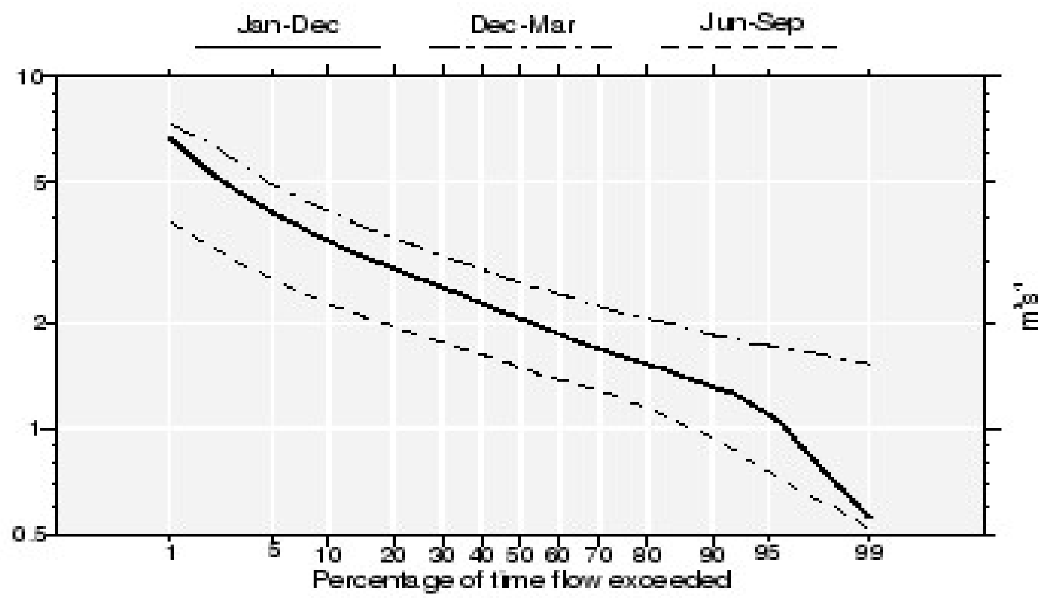


Figure 4.1.1: Flow duration curve for the River Bure at Horstead Mill (1974 – 2000) (NRFA, 2005)

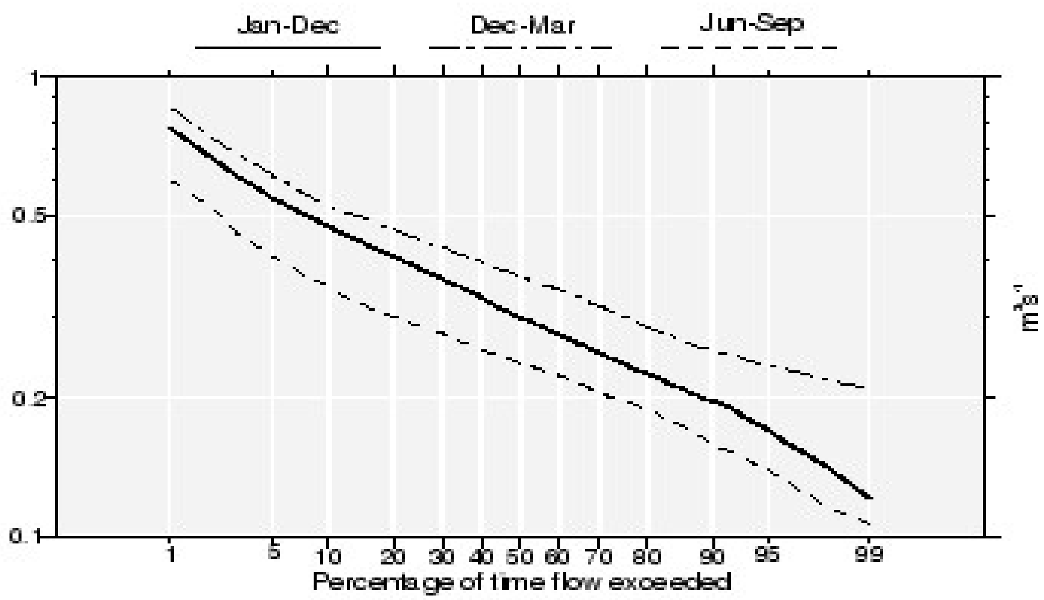


Figure 4.1.2: Flow duration curve for the River Ant at Honing Lock (1966 – 2004) (NRFA, 2005)

The hydrographs in Figure 4.1.3 illustrate a broadly seasonal pattern in flow. Both rivers show a relatively simple regime having one period of high water and one period of low water each year. These periods coincide with high run-off values occurring in the winter months when evaporation is small and peak evaporation during the summer months (Fig 4.1.4). Higher run-off in winter months results in high nutrient run-off in the winter months as shown in Table 4.0.2, where maximum run-off can reach 272.4 mm yr^{-1} in the Bure and 221.5 mm yr^{-1} in the Ant. Nitrate leaching losses are greatest in the winter when plant uptake is minimal, and lowest in the summer when the plants are growing rapidly. The annual infiltration rate of the Bure is approximately 143 mm, and therefore has relatively permeable soils to rainfall, this is similar to the River Ant. Along with their relatively large groundwater component this gives the Bure and Ant a stable flow regime.

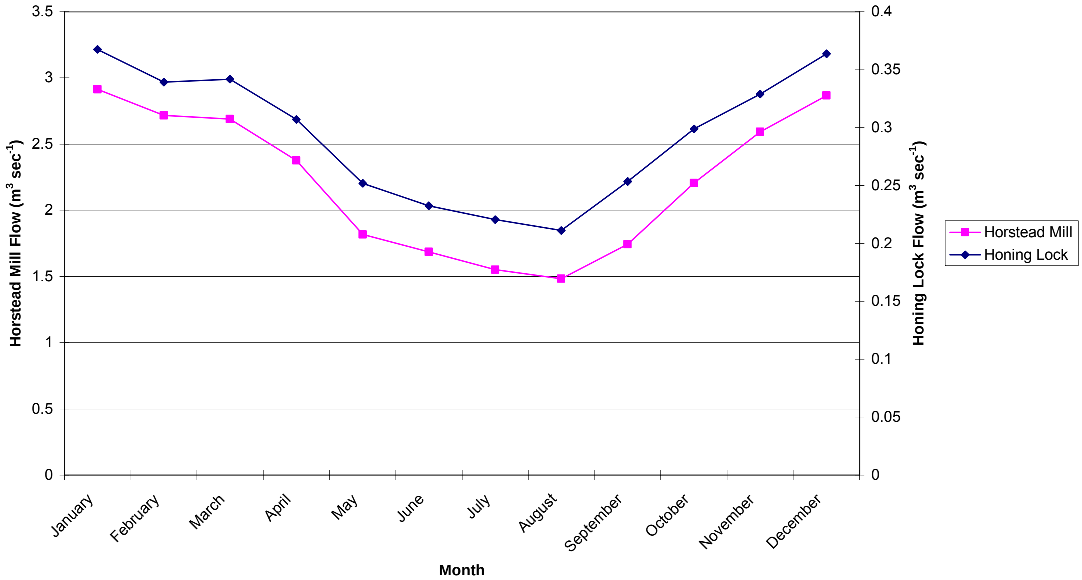


Figure 4.1.3: Average monthly flow at Horstead Mill and Honing Lock (1990-2000)

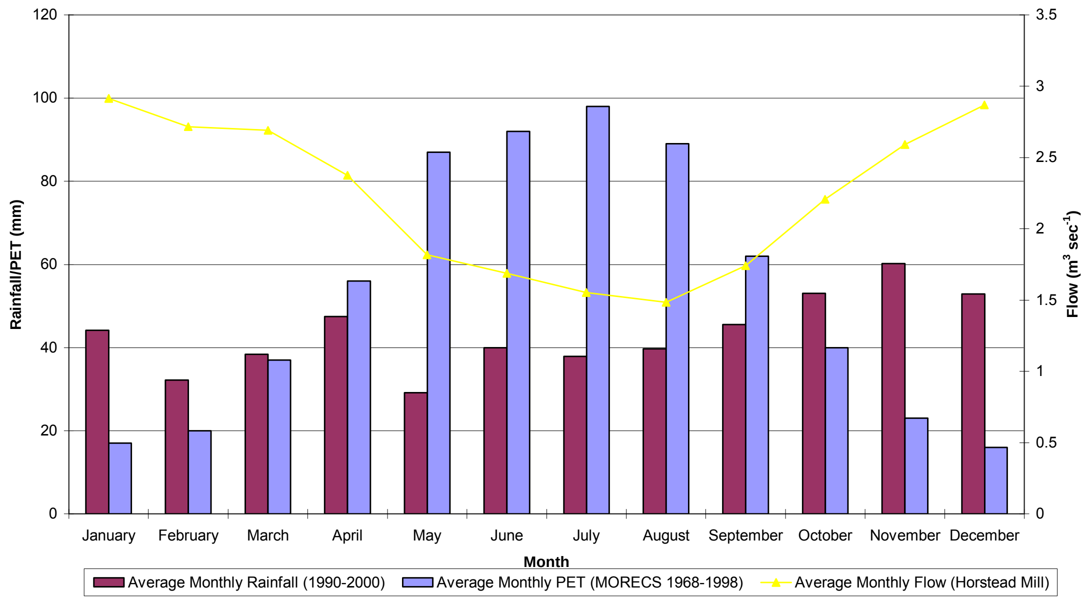


Figure 4.1.4: Average monthly rainfall and PET at Coltishall (River Bure)

In terms of long term variations in flow it can be seen in Figure 4.1.5 that in both rivers there has been an increase in flow over time.

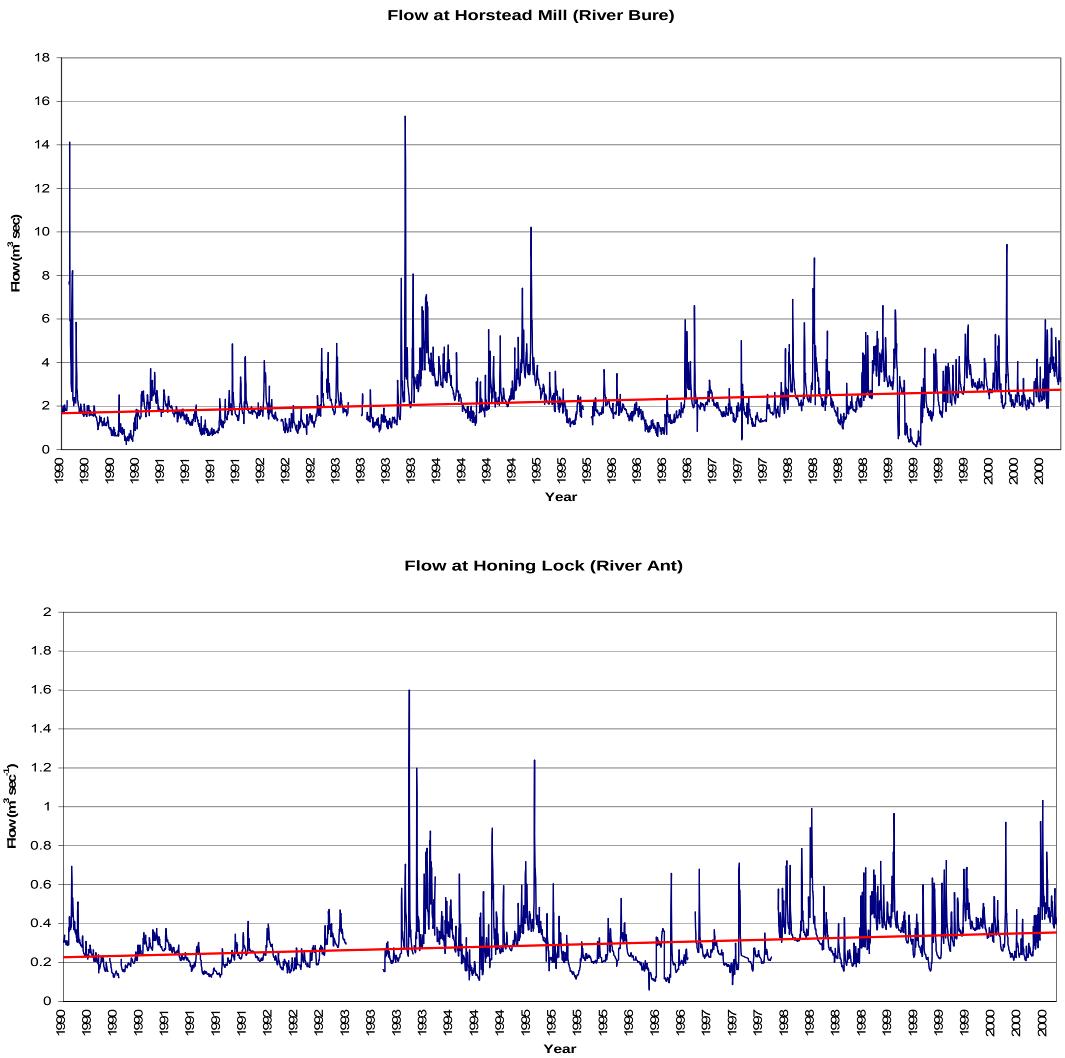


Figure 4.1.5: Variation in flow over time at Horstead Mill and Honing Lock

It can be seen from Figure 4.1.6 that the higher peak flows occurring in 1991, 1994 and again in 1998-2000 can be attributed to higher then average rainfall. Within the study time period 1990-2000 it can also be seen that there are a number of years with above average rainfall and a number of years with below average, however there is only one year with average rainfall (1993). It is know from previous studies that models are unlikely to perform well if calibrated against just particularly dry or wet years. Therefore by using the period 1990-2000 the model will be calibrated and validated against both wet and dry years.

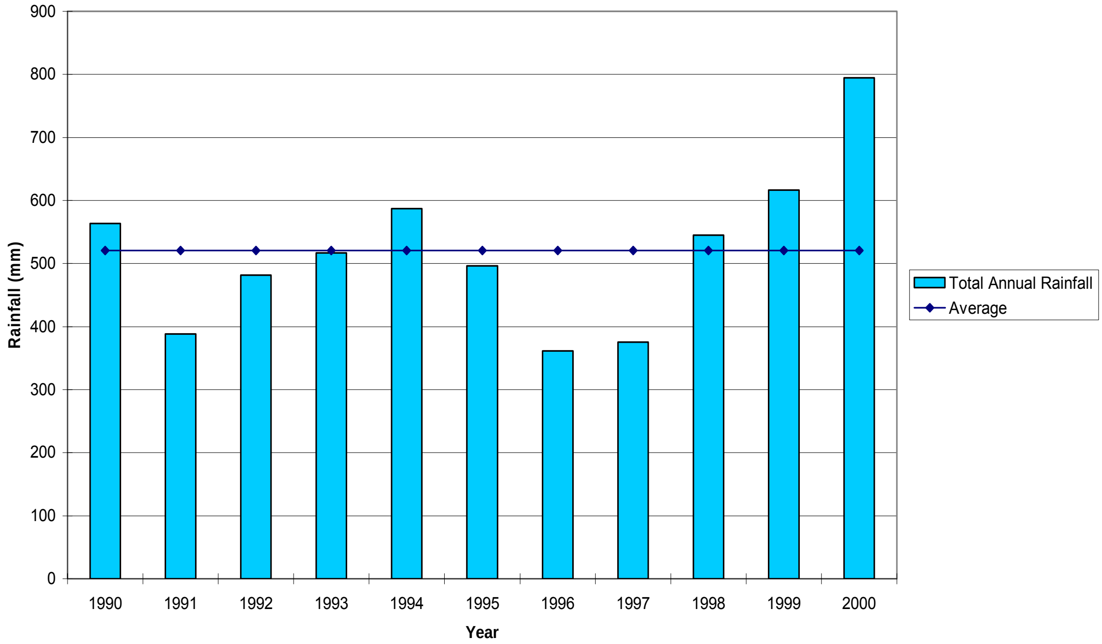


Figure 4.1.6: Average annual rainfall in the study area (1990 – 2000)

Both rivers have high Base Flow Index's (BFI) as can be seen in Table 4.1.1. However total base flow contribution does vary over the year (Figure 4.1.7). It can be seen that throughout the year base flow makes up a considerable amount of total flow in both rivers, especially in the River Ant where it makes up nearly 90% of the flow. The high BFI suggests that most of the water drains down through the soils to a groundwater compartment and on average only 14% in the Ant and 17% of the water flows overland or as near-surface quickflow.

Table 4.1.1: Base flow contribution to total flow

Gauge	BFI (HOST)
Horstead Mill	83%
Honing Lock	86%

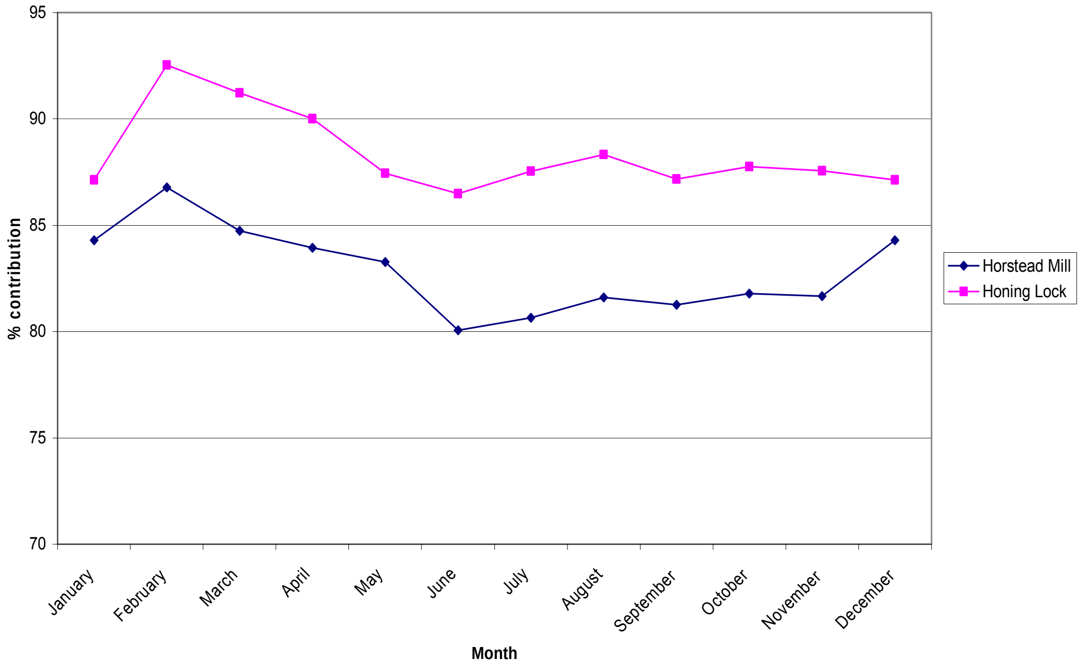


Figure 4.1.7: Monthly average base flow contribution to total flow (1990 – 2000)

Approximately 50% of base flow within the system is made up of STW discharges (Figure 4.1.8). It can also be seen that STW's often discharge more than their consented dry weather flow (DWF) and therefore contribute a significant amount to peak river flows during wet periods, diluting in stream nutrient concentrations. Considering the input of STW discharge into the rivers, especially during the summer months, estimates of base flow to the rivers would be halved. Base flow concentration of nutrients, especially phosphorus, is however likely to be higher during the summer, as can be seen at Horstead Mill (River Ant) (Table 4.0.2). At this time base flow is likely to be composed of water that has been in contact with mineral material for some time and as a result has a higher solute concentration than flows in storm periods, which have had a much shorter period of contact with the vegetation and soil material.

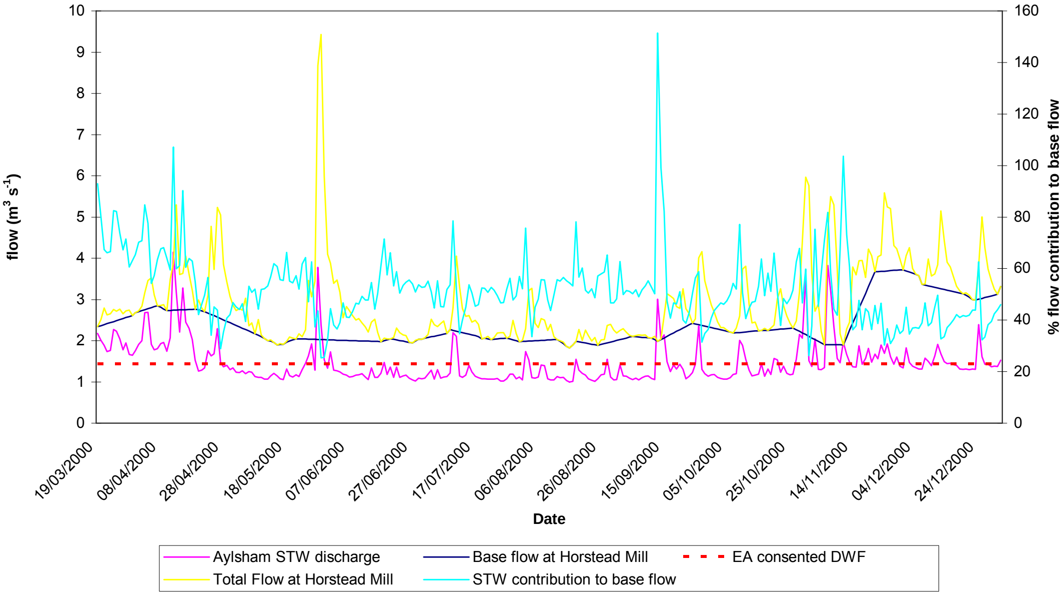


Figure 4.1.8: STW discharge contribution to total and base flow at Horstead Mill

The channels of the River Ant and Bure have an extremely low gradient of 0.16% and 0.17% respectively. This will promote the settlement on to the channel bed of much of the sediment load exported to the channel in high flow events. With this will be the sediment-associated phosphorus load exported from catchment sources (Johnes *et al.*, 1996). The hydrological function of the River Ant and Bure might be expected, therefore, to lead to the storage of sediments and particulate phosphorus within the channel bed sediments under base flow conditions, with re-suspension of the finer fractions during storm events.

The hydrological function of the study area, its topography and soil characteristics provide a good indication of the patterns, which might be generated in the mobilization, and transport of nutrients from diffuse sources in the catchment. However, there are wide ranges of point sources discharging nutrients to the system and the impact of these discharges on the dynamic in stream hydrochemistry cannot be predicted from these broad catchment descriptors.

4.1.2 Initial Water Quality Load Analysis

No flow data are available for the River Thurne so no nutrient loading calculations have been done for this catchment. Nutrient load and flow data for Honing Lock on the River Ant and Horstead Mill on the River Bure for 1990-1999 are presented graphically in Appendix Two, to establish a correlation between flow and nutrient load. These graphs were inconclusive so correlation coefficients have been calculated. The significance of the correlation coefficient has been tested using the student T test (Table 4.1.2).

Table 4.1.2: Significance of the correlation coefficients

Site	Nutrient	Sample size (n)	Correlation Coefficient (r)	T cal.	T tab.	Significance at 5 % level
Horstead	N	48	0.255	1.79	1.68	Significant
	P	22	0.467	0.21	1.72	Not significant
Honing	N	12	0.030	0.09	1.81	Not significant
	P	24	0.037	0.17	1.72	Not significant

Only nitrogen loads at Horstead Mill show a significant correlation to flow. From the graph in Appendix Two it can be seen that, as expected, nitrogen loading increases with high flow due to storm events and increased run-off.

4.1.3 Initial Broads Water Quality Analysis

Appendix Three shows the monthly mean Total phosphorus and Total Oxidised Nitrogen concentrations in the Upper Thurne Broads from 1978 - 2001.

Hickling and Horsey Mere show a similar annual pattern with lower TP concentrations at the start of the year, increasing very slightly to a small peak in September then decreasing again. This pattern is much more distinctive in Heigham Sound, Candle Dyke and Martham Ferry. They all have peak TP concentrations during July and August, with lowest concentrations from December to March. Martham North and South Broad have matching patterns of steady low concentrations, with minimum concentrations occurring in September.

The mid channel sample points, Heigham Sound, Candle Dyke and Martham Ferry, have lower TP concentrations during the winter and early spring due partly to the higher discharge in the channels at this time of the year. Increased volume of water dilutes the amount of P present, to give a lower concentration in the water sample. Figure 4.1.9 shows the water levels during the winter months in Hickling. Although the Broad itself does not change markedly in TP concentration throughout the year, the water flowing in the narrower channels does. Sediment re-suspension may cause the higher TP concentrations in the summer channel sample points, as boat traffic is the highest during these months (Hoare, 2002).

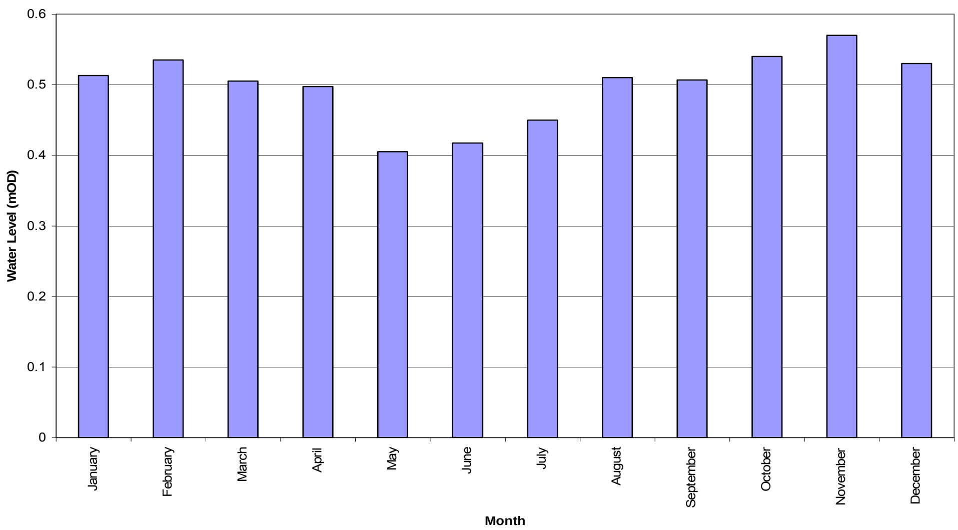


Figure 4.1.9: Monthly mean water level from Hickling Broad (1993 – 2000)

In terms of TON concentrations all the sample sites show the same annual pattern with higher concentrations in the winter, peaking in February, this suggests that TON concentrations are a result of groundwater contribution to the system. From February there is a decline in the concentrations to a minimum value in August.

4.1.4 Initial Pump Loading Analysis

From the electrical consumptions of the IDB pumps monthly mean discharge rates have been calculated as m^3 day^{-1} by multiplying given conversion factors by electricity consumption rates. The following conversion factors were used for each pump:

Table 4.1.4: Conversion factors used for calculating water volume (m^3) discharged by the IDB pumps from electricity consumption (kWh) (Holman, 1994)

Pump	Conversion Factor m^3 kWh
Catfield	153
Stubb Mill	83
Eastfield	66
Brograve	44
Horsey Mill	48
Somerton North	51
Somerton South	63

Results of water quality monitoring from the IDP pumps are most usefully displayed in terms of the load delivered to the system. This represents how much actual mass of a particular substance has entered the system in a given period. In this case a mean rate has been calculated as kg day^{-1} for each month sampled. Multiplying the mean monthly concentration of the substance by the total monthly discharge volume over the number of days in the month derived the daily rate.

There were several years when water quality data and pump discharge data were either not collected or not recorded. The graphs in Figures 4.1.10 and 4.1.11 represent mean daily loads of Total Phosphorus (kg day^{-1}) and Total Oxidised Nitrogen (kg day^{-1}) for the IDB drainage levels for all collected data.

For all the pumps winter is the period of highest P discharge with January and December having the highest loadings, suggesting that concentrations are a result of surface run-off. The pumped volume data also shows the winter as the period of greatest water discharge rate. July and August have the lowest TP loading, along with the lowest water discharge rate.

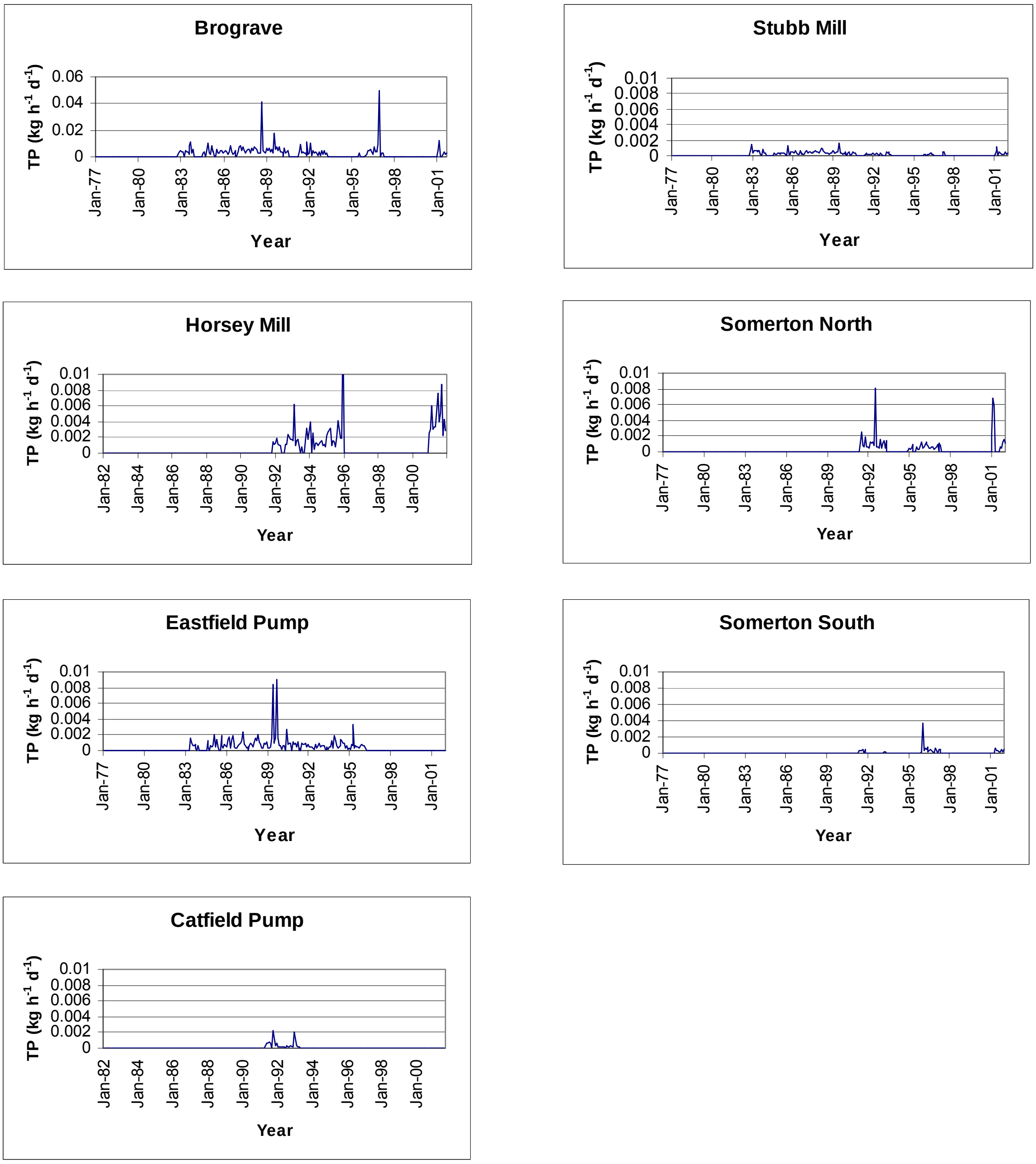


Figure 4.1.10: Graph of mean Total Phosphorous loads delivered by the IDB pumps in the Upper Thurne

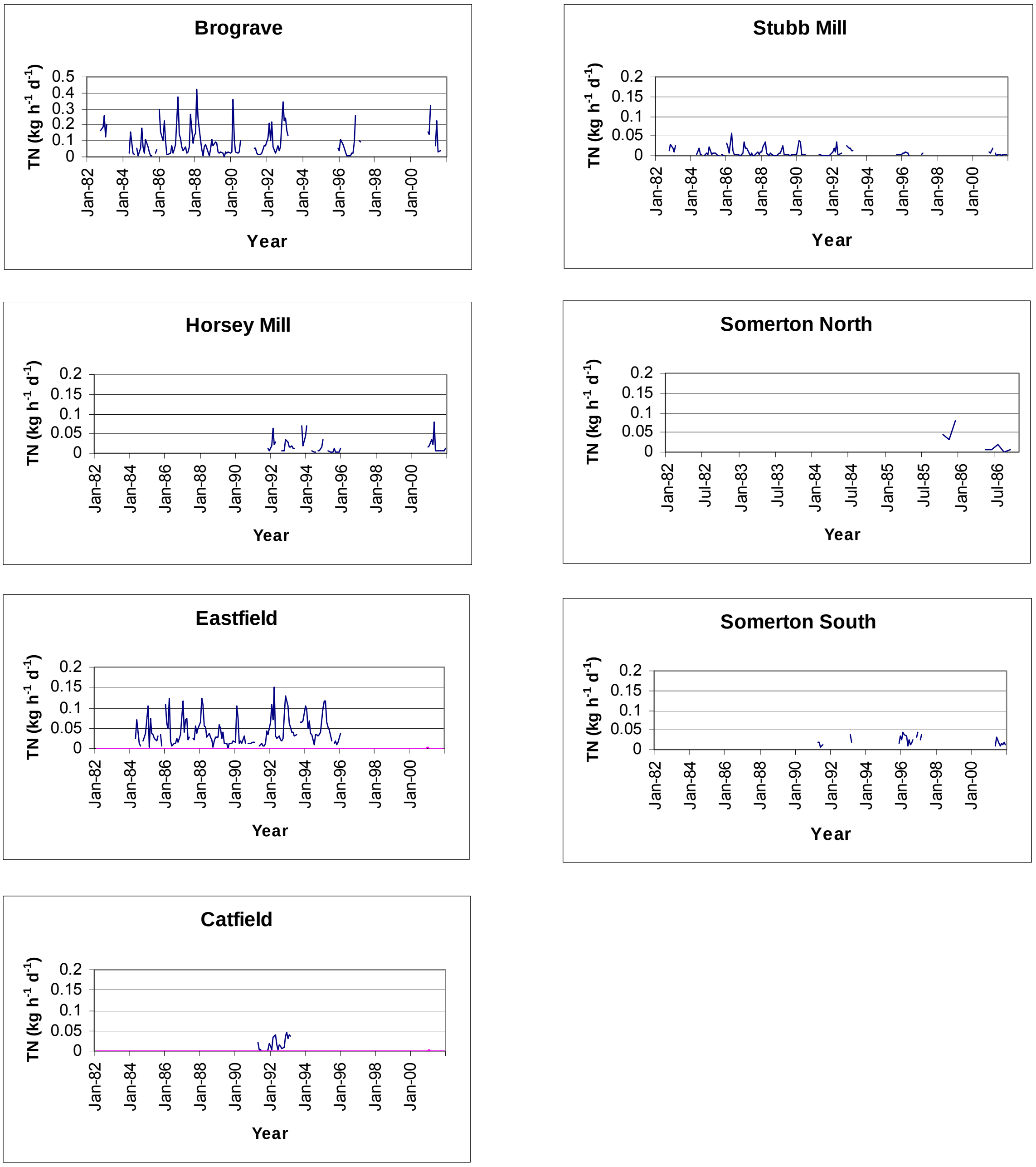


Figure 4.1.11: Graph of mean Total Oxidised Nitrogen loads delivered by the IDB pumps in the Upper Thurne

TON loads also tend to be higher in the winter with peak loadings occurring in January and February, coinciding with high pump discharge rates.

The data suggest that TP loads from the catchment have remained relatively consistent in the period from 1982 – 2001. However higher peak values can be seen over the last 10 years especially at the Horsey Mere and Brograve pumps. Higher peak TON loads can also be seen at Brograve in 2001. Peak TON loadings also occurred at Eastfield Pump, Brograve and Stubb Mill in 1984. The majority of these peak loading concentrations can be attributed to high discharge rates for example the high TP loading occurring at Brograve in June 2001 ties in with the highest discharge rate in the whole data series (88107.9 m^3 day^{-1}) (see Appendix Four for pump discharge rates). However the peak TON loading at Brograve pump in 1984 cannot be accounted for by high discharge levels, it is the actual TON concentration which is high at 17.60 mg l^{-1}.

An overall increase in discharge rates at the Brograve pump from 2001 can account for the more frequent higher TP and TON loadings. This increase may have occurred due to either unusually heavy rainfall or because of the diversion of Eastfield water to the Brograve pump, increasing the agricultural area drained by Brograve Pump. This diversion occurred some time in 1996; however nutrient data are unavailable for 1997 – 2000 so it is difficult to confirm that this may have been the cause of the higher discharge rate (Table 4.1.5).

Table 4.1.5: Brograve average monthly pump consumption (Kwh)

Year	Average monthly pump consumption (Kwh)
1995	2041.5
1996	6225.3
1997	5661.5
1998	13107.0
1999	15726.4
2000	12787.3
2001	23654.4

Both the TON loadings of the pumps and concentrations found in the Broads follow a similar annual pattern with higher values occurring in the winter. However this is not the case with TP. TP concentrations are higher in the summer months in the Broads when the summer pump loadings are at their lowest. This is attributed to the internal TP loading of the broads, which is released from the sediment as the water temperature increases. It is therefore clear that the discharge rate of the pumps play an important role in determining the nutrient loading entering the Upper Thurne system.

4.2 Implications for Modelling

The above analysis of water quality and flow data has highlighted a number of areas within the system which are important to 'get right' when modelling as they impact on a number of other processes. Nutrient load and concentration outputs depend on a number of inputs, e.g. soil data, fertiliser data, climate data, land use and management but more importantly the accuracy of the modelled system's hydrology which is also dependent on the above mentioned inputs. Therefore the quality of input data is a major component and limitation of the model. Limitations of various data and the implication to the modelling of the system can be seen in the table below.

Table 4.2.0: Implications of data input to the modelling of the system

Data Input	Data quality/ availability	Impact on system and modelling	Solution
Rainfall	Daily data are available from BADC but many missing values and data are inaccurate by a factor of 10	Important to model flows correctly, run-off contribution to flow, groundwater and evapotranspiration. Incorrect modelling will affect nutrient loads.	A computer program has been devised by other BADC users, which will infill data gaps and correct data by a factor of 10. Discrepancies in data will still occur.
PET	No available data, only long term averages are available	Daily data needed for modelling. Will impact hydrology of the system	Can be calculated either by the model or outside the model. Investigation into best method is required.
IDB pumps	Sparse electrical data are available for the pumps.	Cannot model pumps within SWAT. IDB pumps are an important feature in the Thurne system in terms of hydrology and water quality	Electrical data can be converted to pumped volumes as seen in section 4.1.4. Investigation into the possible methods of modelling the pumps in SWAT is needed.
Base flow	No daily base flow data are available for calibration of model	Important component for the hydrology of the system. Need to get right to model nutrient loads correctly.	The SWAT user manual suggests a method to calculate base flow using SWAT outputs; however this method may not be used with observed flow data. By using the IH turning point method (section 4.1.1) for both observed and SWAT predicted flow data sound comparisons can be made between the 2 data sets to enable the calibration of base flow.
STW discharge	Few data are available for this. There are only water quality limits available for 3 sites. Daily discharge data are only available for 2 sites within the system – but only for 2000.	STW discharge is a very important part of the system as it makes up over 50% of the base flow. It also impacts on the wet weather flow, making up a large amount of the peak flow.	Consented DWF is available for all sites but as can be seen in the analysis during wet periods STW makes up a high proportion peak flow, therefore extra flow will be missed from the system. This will result in lower modelled peak flows.

Chapter Five Catchment Scale Modelling

5.0 The international use of the Soil Water Assessment Tool (SWAT)

As was discussed in Chapter Two river basin modelling using the Soil and Water Assessment Tool (SWAT) is becoming widely adopted throughout the world for addressing different dimensions starting from hydrological modelling to policy making from a small catchment to a regional or continental level. Although it was developed for United States conditions it is used in many countries throughout Europe, including assessment of the model for use in the implementation of the European Union Water Framework Directive (Dilks *et al*., 2003). The SWAT model is included in another European project intended for policy making. EUROHARP (EUROpean HARmonised Procedures) is a project aimed at developing European harmonised procedures for the quantification of nutrient losses from diffuse sources (http://www.euroharp.org/pd/pd/index.htm).

The above studies discussed the selection of SWAT for projects at a European scale. At the river basin scale, Shepard *et al*., (1999) conducted an investigation to select suitable models for modelling nutrient transport to watercourses in the UK. Their evaluation revealed SWAT as the best package for assessing river basin scale nutrient pollution in the UK. The above mentioned studies have rigorously assessed SWAT before using the model to address their problems. Studies of this kind give direction to others who are in the process of selecting a modelling tool for investigating a specific environmental problem. These studies also support the selection of SWAT for use in the UK and in the context of this research.

5.1 Soil Water Assessment Tool (SWAT)

SWAT is the acronym for Soil and Water Assessment Tool, it is a river basin, or watershed, scale model developed by Dr. Jeff Arnold for the USDA Agricultural Research Service (ARS) (USDA Agricultural Research Service, 2000). SWAT is a semi distributed model which is able to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex watersheds with varying soils, land use and management conditions over long periods of time.

The benefits of using SWAT for this study are:

- A semi-distributed or distributed approach allows the spatial interactions between weather, soil and land-use to be simulated; source areas of nutrient runoff can be identified to enable simulation and planning of future land management controls. Remote sensing techniques may be a valuable tool for acquiring both spatial and temporal cropping / land management data.

- SWAT is a mechanistic model, in which the physical processes are represented in some explicit form, meaning that short term dynamics and management controls can be simulated.

5.2 Data Collection

The following is a summary of data required to set up a SWAT model. Data obtained and the sources of data are shown in Appendix Five.

Spatial data

- Topography
- Locations of the stream network, reservoirs and ponds
- Land cover
- Soil type and characteristics

Temporal data

- Daily weather data (rainfall, extreme temperatures, solar radiation, wind speed, relative humidity, potential evapotranspiration)
- Agricultural practices (fertiliser and pesticide application, tillage, crop rotation schemes)
- Water abstractions and discharges

- Land use statistics

Calibration/validation data

- River discharge
- Water quality

5.3 Model Build

The initial model build exercise has concentrated on the Bure and Ant river basins only as these are simpler to model than the Thurne. Calibration parameters used for this initial model can then be used as a starting point for the Thurne model. The model has initially been built for the year 1990 for calibration purposes due to data availability restrictions.

Soil and crop type data are only available at a national, regional level or parish level. These data are supplied in grid format with an associated dominant crop type or soil group, and a range of sub-dominant classes. In this context dominant is used in a spatial sense rather than a hydrological one, but sometimes a soil or crop, which covers less of the area, is more important in controlling either hydrological or erosional response, or both. As this study is focussed on potential future conditions it is essential that past and current conditions be modelled as accurately as possible, and that the responses at the sub-basin and basin level actually reflect the processes expected. SWAT is a comprehensive model that requires a diversity of information in order to run; therefore great care has been taken in selecting data for use in the model set up.

5.3.1 Sub-basin Delineation

The first step in setting up a river basin simulation is to define the configuration of the river basin. A detailed 50m grid, digital elevation model (DEM) supplied by CEH-Wallingford has been used to ascertain the river basin outline of the Bure and Ant river basin. Due to the low elevation and gradient of the river basins and SWAT's inability to model ground levels below sea level, 10m has been added to all ground levels (Fig 5.3.1) within the model.

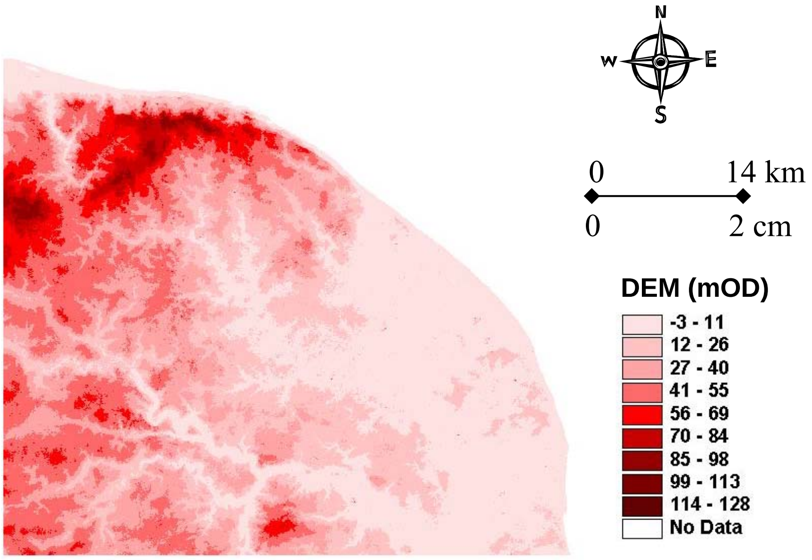


Figure 5.3.1: CEH Wallingford DEM of the Norfolk region

The river network for the study area was also supplied by CEH – Wallingford and was used as an aid in the automatic definition of the river network within SWAT. The locations of EA gauging stations were used to help define outlets for sub-basins along with water quality sampling sites to help later with calibration and validation exercises.

The Bure and Ant has been delineated into 29 sub-basins (Fig 5.3.2). Sub-basins possess a geographic position in the watershed and are spatially related to each other. For example, from Figure 5.3.2 it can be seen that outflow from sub-basin 23 enters sub-basin 22. These sub-basins are defined by the surface topography provided by the DEM so that the entire area within a sub-basin flows to the sub-basin outlet. A sub-basin will contain at least one hydrological response unit, a tributary and a main channel or reach.

Hydrological response units (HRUs) are portions of a sub-basin that possess unique combinations of land use/management and soil attributes. Whilst individual fields with a specific land use, management and soil may be scattered throughout a sub-basin SWAT will lump these areas together to form one HRU. This process will simplify a SWAT run, as it is often not practical to simulate individual fields. Within the Bure and Ant model each sub-basin has between 1 and 5 HRU's.

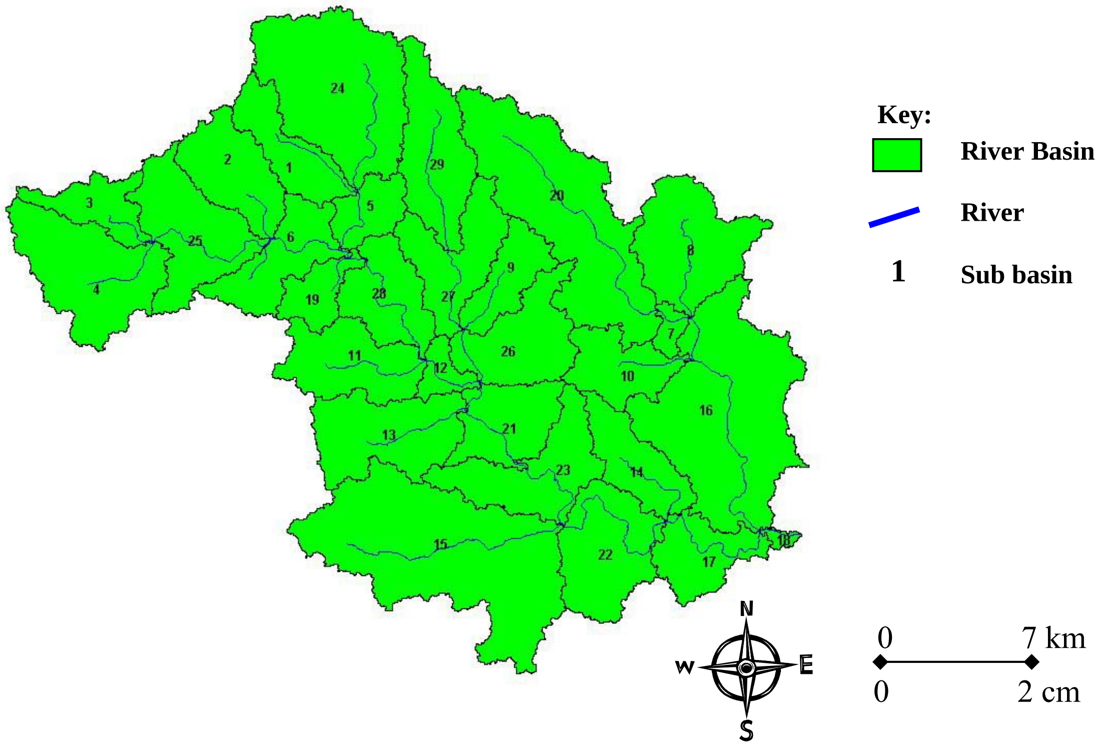


Figure 5.3.2: SWAT study area and sub basins

5.3.2 Meteorological Data

SWAT requires daily precipitation, temperature, relative humidity, wind speed and solar radiation data. Values may be read from records of observed data or may be generated. Daily precipitation, temperature and wind speed data were acquired from the British Atmospheric Data Centre (BADC). Relative humidity was then calculated from wet and dry bulb temperatures and solar radiation was estimated from hours of sunshine as can be seen in the equations below.

Relative humidity (%) $RH = 100\,\frac{e_d}{es_{db}}$

Where:	es_{wb}	saturation vapour pressure at T_{wb} (kPa)
	es_{db}	saturation vapour pressure at T_{db} (kPa)
	e_d	vapour pressure (kPa)
	Elv	elevation above sea level (m)
	P	air pressure (kPa)

T_{wb} wet bulb temperature (°C)

T_{db} dry bulb temperature (°C)

Approximate the air pressure, P in kPa (kiloPascals):

$$P = 101.325 \exp(-0.0001184\, Elv)$$

Conversion factor, A:

$$A = 0.00066(1.0 + 0.00115 T_{wb})$$

Saturation vapour pressure at T_{wb}.

$$es_{wb} = \exp\left(\frac{16.78 T_{wb} - 116.9}{T_{wb} + 237.3}\right)$$

Vapour pressure, or the partial pressure of water vapour, e_d in kPa:

$$e_d - es_{wb} - AP(T_{db} - T_{wb})$$

Calculate the saturated vapour pressure, es_{db}:

$$es_{db} = \exp\left(\frac{16.78 T_{db} - 116.9}{T_{db} + 237.3}\right)$$

Solar radiation was calculated using the Angstrom formula, which relates solar radiation to extraterrestrial radiation and relative sunshine duration:

Solar radiation (R_s) $R_s = \left(a_s + b_s \frac{n}{N}\right) R_a$

Where: R_s solar or shortwave radiation (MJ m^{-2} day^{-1})

N actual duration of sunshine (hour)

N maximum possible duration of sunshine or daylight hours (hour)

n/N relative sunshine duration

R_a extraterrestrial radiation (MJ m^{-2} day^{-1})

a_s	regression constant, expressing the fraction of extraterrestrial radiation reaching the earth on overcast days (n = 0)
a_s+b_s	fraction of extraterrestrial radiation reaching the earth on clear days (n = N).

R_s is expressed in the above equation in MJ m^{-2} day^{-1}. The corresponding equivalent evaporation in mm day^{-1} is obtained by multiplying R_s by 0.408. Depending on atmospheric conditions (humidity, dust) and solar declination (latitude and month), the Angstrom values a_s and b_s will vary. As no actual solar radiation data are available the recommended values $a_s = 0.25$ and $b_s = 0.50$ where used. Values for R_a (16.83 MJ m^{-2} day^{-1}) and N (11.98 hours) were based on the latitude 52.69° for Coltishall (Allen *et al.,* 1998). The actual duration of sunshine, n, was taken from BADC data.

Data from a total of 16 rain gauges and 3 temperature gauges (Coltishall, Hemsby and Melton Constable) were collated for the three river basins (Bure, Ant and Thurne), these gauges were selected as they had the most recent available data. Only 10 rain gauges were located within the Bure and Ant river basins, of these only 7 were used due to lack of data at Stalham (only 2 years of data in total), Buxton Dudwick Cottage (only data from 1994) and Hevingham (only data up to 1997) (Fig 5.3.4).

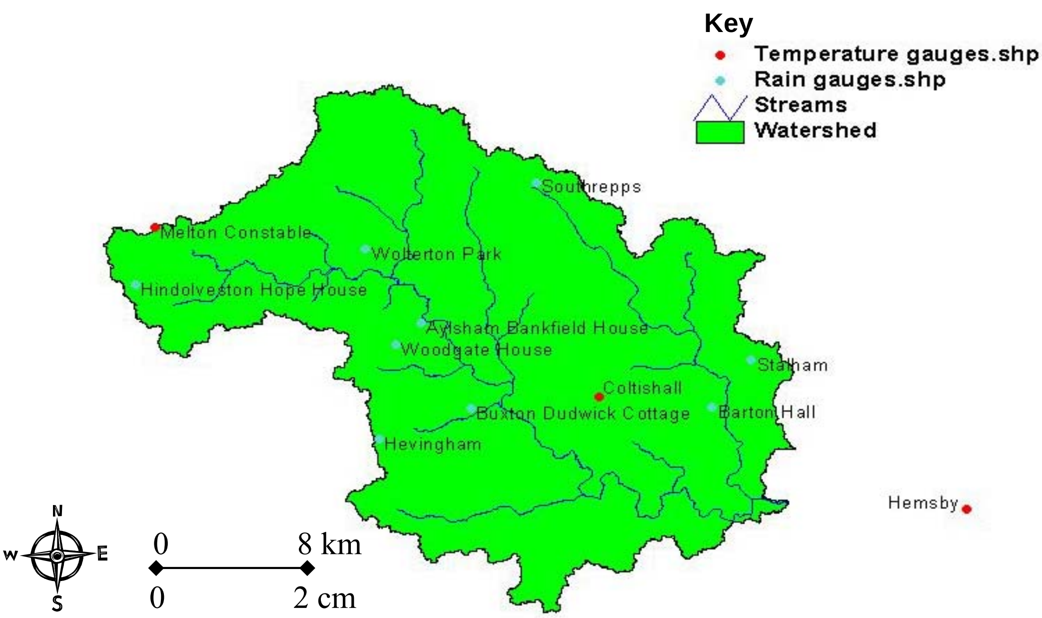


Figure 5.3.4: BADC rain gauge locations used in the model

Any missing values in the data set were infilled by deriving a relationship with Aylsham Bankfield House. This gauge was selected, as it was closest to the areal average rainfall. This was for both the Arithmetic Mean and Thiessen Polygon method, (Appendix Six) within the Bure and Ant river basin.

Only the Coltishall temperature gauge has been used, as it was the only gauge located within the Bure and Ant river basin that had all the required data. The gauge located at Hembsy also had all the relevant data but is within the Thurne river basin.

5.3.3 Evapotranspiration

Daily potential evapotranspiration values are required by SWAT. These can be either calculated by SWAT or read in by the user. The BADC data has also been used to calculate potential evapotranspiration. Three methods have been used:

Penman-Monteith
$$\lambda E = \frac{\Delta \cdot (H_{net} - G) + \rho_{air} \cdot c_p \cdot (e^o_z - e_z)/r_a}{\Delta + \gamma \cdot (1 + r_c / r_a)}$$

Where λE is the latent heat flux density (MJ m^{-2} d^{-1}), E is the depth rate evaporation (mm d^{-1}), Δ is the slope of the saturation vapor pressure-temperature curve, de/dT (kPa $^oC^{-1}$), H_{net} is the net radiation (MJ m^{-2} d^{-1}), G is the heat flux density to the ground (MJ m^{-2} d^{-1}), ρ_{air} is the air density (kg m^{-3}), c_p is the specific heat at constant pressure (MJ kg^{-1} $^oC^{-1}$), e^o_z is the saturation vapor pressure of air at height z (kPa), ez is the water vapor pressure of air at height z (kPa), γ is the psychrometric constant (kPa $^oC^{-1}$), r_c is the plant canopy resistance (s m^{-1}), and r_a is the diffusion resistance of the air layer (aerodynamic resistance) (s m^{-1}).

Hargreaves
$$\lambda E_o = 0.0023 \cdot H_0 \cdot (T_{mx} - T_{mn})^{0.5} \cdot (T_{av} + 17.8)$$

Where λ is the latent heat of vaporization (MJ kg^{-1}), E_o is the potential evapotranspiration (mm d^{-1}), H_0 is the extraterrestrial radiation (MJ m^{-2} d^{-1}), T_{mx} is the maximum air temperature for a given day (°C), T_{mn} is the minimum air temperature for a given day (°C), and Tav is the mean air temperature for a given day (°C).

Priestley-Taylor $$\lambda E = \alpha_{pet} \cdot \frac{\Delta}{\Delta + \gamma} \cdot (H_{net} - G)$$

Where λ is the latent heat of vaporization (MJ kg^{-1}), E_o is the potential evapotranspiration (mm d^{-1}), α_{pet} is a coefficient, Δ is the slope of the saturation vapor pressure-temperature curve, *de/dT* (kPa $^oC^{-1}$), γ is the psychrometric constant (kPa $^oC^{-1}$), H_{net} is the net radiation (MJ m^{-2} d^{-1}), and G is the heat flux density to the ground (MJ m^{-2} d^{-1}).

Figure 5.3.5 shows the monthly PET (potential evapotranspiration) values for 1990 for all three methods. These have been compared against areal average values given by Smith (1976) based on measured data for the period 1941 – 1970 and MORECS for the region (1961 – 1998). The Hargreaves method compares well to MORECS and Smith (1976) data throughout the year. The Priestley-Taylor method over estimates PET values throughout the year as does the Penman-Monteith method but not to the same magnitude. The values provided by Smith (1976) are lower for the winter months than those shown by the MORECS data. This may because of climate change since the 1970's, which has resulted in a shift in PET values for the area.

These values can later be compared to PET estimates given by SWAT using the Hargreaves method. The values estimated by SWAT will affect the model's ability to model flow volumes leaving any part of the model. Consequently the accuracy of the estimated values against local data is important. If SWAT significantly under or over estimates PET then values calculated outside of SWAT using the above Hargreaves method can be fed into the model as an 'observed' PET data set.

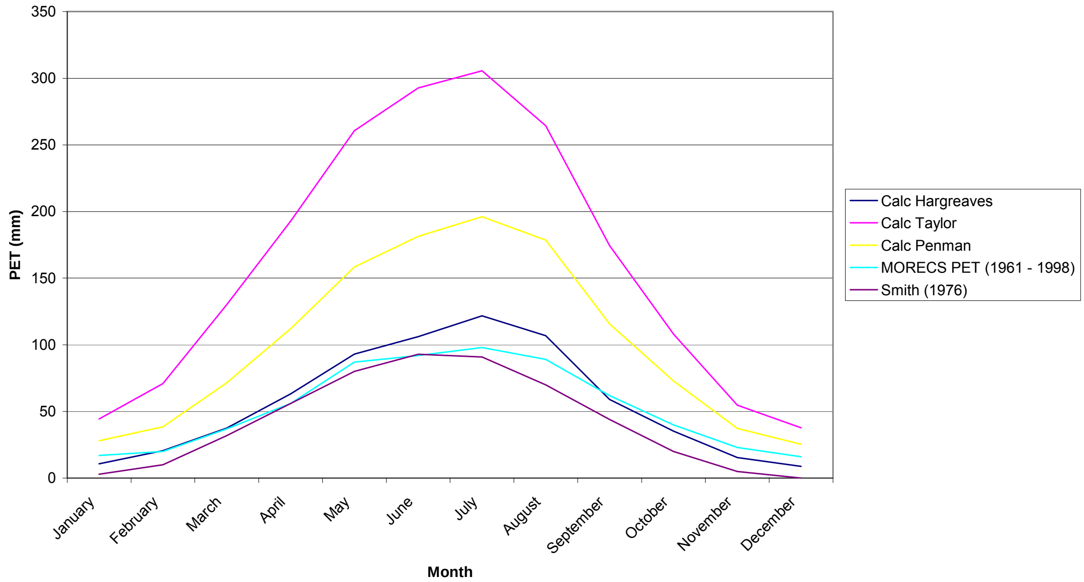


Figure 5.3.5: Evapotranspiration Comparisons

5.3.4 Land Use

The CEH 1990 land use map provided spatial distribution of major land cover classes within the study area (Fig 5.3.6).

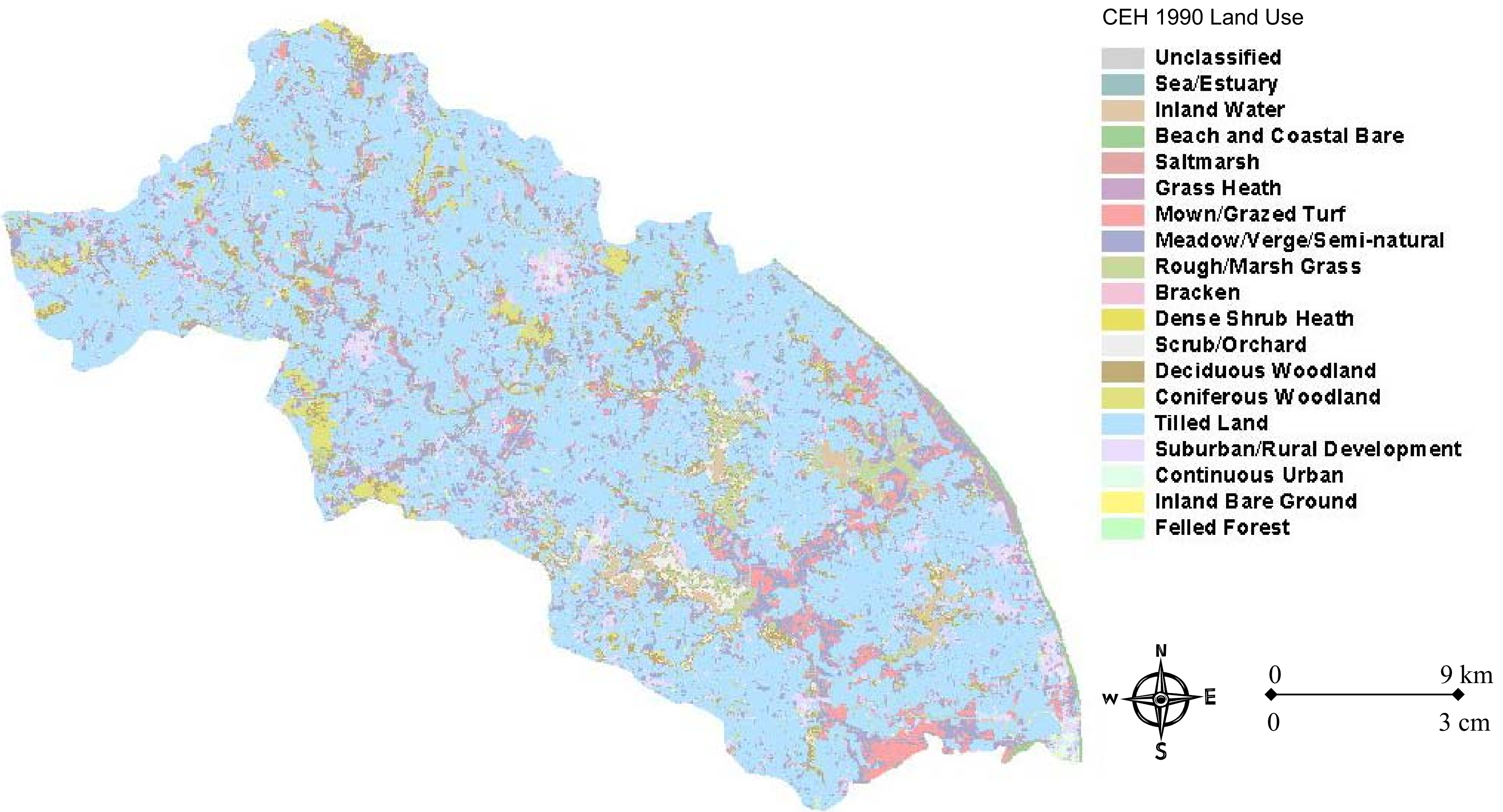


Figure 5.3.6: 1990 land use map for the whole study area

This land use map however defines all arable land within one class; as a result Edinburgh Data Library Agricultural Census data have also been obtained. Data have been collated for a 2km grid resolution for 1969, 1976, 1981, 1988, 1994, 1997 and 2000. These two map layers have been overlain and for every intersection of EDL (1994, 1997 and 2000) and 1990 land use map areas defined as arable the dominant crop from EDL has then been used to help define crop rotations for the study area (section 5.3.6).

5.3.5 Soils

The soil data used by SWAT can be divided into two groups, physical characteristics and chemical characteristics. The physical properties of the soil govern the movement of water and air through the profile and have a major impact on the cycling of water within the HRU. Inputs for chemical characteristics are used to set initial levels of the different chemicals in the soil. While the physical properties are required, information on chemical properties is optional. The SWAT soil input file defines the physical properties for all the layers in the soil, so is very data intensive.

The data required encompasses a large number of paper soil reports, maps and a large number of digital soil information. The NSRI LandIS database incorporates all these data sources and has been utilised to gain the appropriate data for SWAT. LandIS data are based on an average of 2 to 3 soil observations nationally. The National Soil Map details the distribution of 300 soil associations each of which contains three to five soil series. Together, these soil associations describe the wide range of soil conditions encountered across England and Wales. The average national percentage composition of each soil association is estimated from field experience. In these aggregated datasets, each mapped area is described according to the predicted percentage soil series composition. Therefore soil data provided by LandIS does not represent soils on a local scale but is based on a small number of national samples and predicted values. This may affect the accuracy of the soil modeling within SWAT and the sensitivity of soil associations and soil series is investigated further in this section.

Some of the soil data SWAT required does not come directly from LandIS but was calculated using LandIS values and equations taken from the TERRACE project

(White *et al.,* 2002). A spreadsheet was therefore developed which can be used for this and other projects, listing SWAT data requirements and the relevant LandIS data parameters needed (Table 5.3.1).

Table 5.3.1: SWAT soil data requirements

SWAT Soils	Description	Source	LandIS Property Needed
Soil Name	Soil Association	LandIS	SERIES_NAME
Number of layers	Number of layers	LandIS	LAYER_DESIGN
Hydrological group	Soil hydrological group (A, B, C,D)	LandIS	HOST, BFI, SPR
Rooting depth	Maximum rooting depth of soil profile	LandIS	DROCK
Anion exclusion factor	Fraction of porosity (void space) from which anons are excluded	Default value 0.50	
Potential crack volume	Potential or maximim crack volume of the soil profile expressed as a fraction of the total soil volume	Optional	
For each layer			
Depth from surface	Depth from the soil surface to bottom of layer	LandIS	UPPER_DEPTH, LOWER_DEPTH
Depth of obstacle to roots	Maximum rooting depth of soil profile	LandIS	DROCK, DGLEY, DIMP_DP
Bulk density	Moist bulk density	LandIS	BULK_DENSITY
Available water content	Available water capacity of the soil layer	LandIS	AWC, THV5, THV1500
Saturated hydraulic conductivity	Saturated hydraulic conductivity	LandIS	KSAT_SV
% organic carbon	Organic carbon content	LandIS	CARBON
% clay	Clay content	LandIS	CLAY
% silt	Silt content	LandIS	SILT
% sand	Sand content	LandIS	SAND_TOTAL
% rock	Rock fragment content	LandIS	
Albedo	Moist soil albedo	Calculated	MATRIX_COLOUR/VALUE
Erodobilty factor	USLE equation soil erodibility (k) factor	Calculated	CARBON, CLAY, SILT, SAND_TOTAL

Soil albedo is a function of soil colour and angle of incidence of the solar radiation, and depends on the inherent colour of the parent material, organic matter content and weathering conditions. The following equation has been used to calculate soil albedo (NRCS, 2005).

Albedo = (0.07 x colour value) – 0.12

The erodibility factor can be calculated using the Universal Soil Loss Equation (USLE). The one used in this research and in SWAT is an alternative equation by Williams (1995):

$$K_{USLE} = f_{csand} \cdot f_{cl-si} \cdot f_{orgc} \cdot f_{hisand}$$

Where f_{csand} is a factor that gives low soil erodibility factors for soils with high coarse-sand contents and high values for soils with little sand, $f_{cl\text{-}si}$ is a factor that gives low soil erodibility factors for soils with high clay to silt ratios, f_{orgc} is a factor that reduces soil erodibility for soils with high organic carbon content, and f_{hisand} is a factor that reduces soil erodibility for soils with extremely high sand contents. The factors are calculated:

$$f_{csand} = \left(0.2 + 0.3 \cdot \exp\left[-0.256 \cdot m_s \cdot \left(1 - \frac{m_{silt}}{100}\right)\right]\right)$$

$$f_{cl-si} = \left(\frac{m_{silt}}{m_c + m_{silt}}\right)^{0.3}$$

$$f_{orgc} = \left(1 - \frac{0.25 \cdot orgC}{orgC + \exp[3.72 - 2.95 \cdot orgC]}\right)$$

$$f_{hisand} = \left(1 - \frac{0.7 \cdot \left(1 - \frac{m_s}{100}\right)}{\left(1 - \frac{m_s}{100}\right) + \exp\left[-5.51 + 22.9 \cdot \left(1 - \frac{m_s}{100}\right)\right]}\right)$$

Where m_s is the percent sand content (0.05-2.00 mm diameter particles), m_{silt} is the percent silt content (0.002-0.05 mm diameter particles), m_c is the percent clay content (< 0.002 mm diameter particles), and *orgC* is the percent organic carbon content of the layer (%).

Soil Series

Soil profile characteristics are used to define soils at four levels in a hierarchical system, general characteristics being used at the highest level to give broad separations and more specific ones at the lower levels to give increasingly precise subdivisions. Soil associations are made up of a number of soil series but are named after the dominant soil series. For example Wick 2 is made up of Wick, Wickmere, Sheringham and Aylsham series (see Table 5.3.2). As mentioned earlier the percentage of soil series which make up soil associations are defined at a national level and could therefore be different at a local level. However no local soil data are available.

Table 5.3.2: Soil Associations within the study area

Soil Association	Areas In SWAT (ha)	Ancillary Subgroups	Proportions (%)	Proportions based on SWAT areas (ha)
Wallasea 1	116.06	813 Wallasea	75	87.05
		814 Newchurch	25	29.02
Newport 4	3787.68	551 Newport	76	2878.63
		631 Redlodge	24	909.04
Isleham 2	1463.04	861 Isleham	31	453.54
		1024 Adventurers	29	424.28
		552 Ollerton	20	292.61
		821 Blackwood	20	292.61
Wick 3	17645.72	541 Wick	61	10763.89
		541 Sheringham	28	4940.80
		551 Newport	11	1941.03
Wick 2	31279.82	541 Wick	38	11886.33
		572 Wickmere	36	11260.74
		541 Sheringham	16	5004.77
		543 Aylsham	10	3127.98
Hanworth	2038.96	871 Hanworth	40	815.58
		831 Sustead	30	611.69
		1024 Adventurers	30	611.69
Gresham	1547.74	711 Gresham	63	975.08
		711 Prolleymoor	21	325.03
		831 Sustead	16	247.64
Beccles 1	1039.76	711 Beccles	65	675.85
		712 Ragdale	35	363.92
Altcar 2	621.58	1022 Altcar	50	310.79
		1024 Adventurers	30	186.47
		1025 Mendham	20	124.32
Felthorpe	1674.01	643 Felthorpe	40	669.60
		642 Lakenheath	27	451.98
		821 Blackwood	33	552.42

As can be seen from Table 5.3.2 each soil series within Wick 2 contributes a percentage to the total soil association, however each series has different characteristics. At the moment the soil database in SWAT is made up of the dominant soil series for each association found in the study area, thus the database doesn't take into consideration the characteristics of the other soil series making up the association. Soils that are highly erodible or behave in a different hydrological fashion could therefore be missed from the model giving unreliable results. To assess this problem USLE calculations have been undertaken for each soil series within each sub -basin within the SWAT model (Appendix Seven).

It can be seen from USLE calculations that soil series such as Sheringham have high a mean annual soil loss (3.46 t ha^{-1}) when crops such as potatoes and sugar beet are grown on it. The Sheringham series makes up approximately 28% of the Wick 3 associations and 16% of the Wick 2 associations and therefore covers approximately 9945 ha of the SWAT river basin. The difference between the Sheringham soil series and the Wick association are nominal (see Hodge *et al.,* 1984). The only difference being the percentage of stones in the soil and therefore it is thought that Sheringham, with no stones within it, will be more erodible than Wick.

The Prolleymoor series that makes up 21% of the Gresham association also has a high mean annual soil loss (3.42 t ha^{-1}) when potatoes or sugar beet are grown on it. When ground is left fallow, as may be the case with land allocated for set-aside, erosion rates can reach as high as 7 t ha^{-1}. This can also be seen with the Newchurch series, which contributes 25% of the Wallesea association. All the HRU's, which have been allocated to this soil association in the SWAT model, have been modelled as set-aside due to the wetness class of the association.

By only modelling the soil associations the above soil series are not incorporated into the model, therefore model results could be underestimating soil erosion. The modelling of the soil subgroups within SWAT does however prove problematic. The SWAT ArcView interface requires a soil map linked to the soil database, to provide spatial information on the soil distribution within the river basin. These maps need to be prepared prior to running the interface. Unfortunately the only digital soil map available is that of the National Soil Map, which only displays soil associations. Neither is any quantitative information available on the spatial distribution of the soil subgroups within each soil association. A sensitivity analysis on the distribution of the soil associations and their subgroups on a small sub-basin within SWAT has been carried out. The model is based on the calibrated Bure and Ant model and the results are discussed in chapter Six.

5.3.6 Management Files

Quantifying the impact of land management and land use on water supply and quality is a primary focus of environmental modelling. SWAT allows very detailed management information to be incorporated into a simulation at the HRU level. The user may define the beginning and the ending of the growing season; specify timing, type and amount of fertiliser, pesticide and irrigation applications as well as timing and type of tillage operations. At the end of the growing season, the biomass may be removed from the HRU as yield or placed on the surface as residue. In addition to these basic management practices, operations such as grazing, automated fertilisation and water applications are available.

Crop Rotations

Crop rotations are a system of regularly changing the crops grown on a piece of land. The crops are grown in a particular order to utilize and add to the nutrients in the soil and to prevent the build-up of insect and fungal pests. Including a legume crop, such as peas or beans, in the rotation helps build up nitrate in the soil, because the roots contain bacteria capable of fixing nitrogen from the air. With a few notable exceptions such as winter wheat and forage maize, most crops are best grown in rotation with other crops. However, there are no mandatory rotations, and no single rotation necessarily represents best practice. Individual farmers will deviate from them to allow for their own machinery / labour availability or personal preferences or because of market prices/subsidies and weather and soil moisture conditions in a given year.

A rotation in SWAT refers to a change in management practices from one year to the next. There is no limit to the number of years of different management operations specified in a rotation. SWAT does not limit the land cover/crops grown within one year in the HRU. However, only one land cover can be growing at any one time.

Originally typical rotations for the eastern region of the UK were used to vary crops grown from year to year within the SWAT model. These were taken from ADAS standard rotation information and based on soil type (Holman, 2004) (Table 5.3.3).

Table 5.3.3: ADAS standard crop rotations (Holman, 2004)

Soil	Rotation	Crops	Soil	Rotation	Crops
Sandy	Primary	pts/ww/sb/sbt/ww/wb	Deep Silty	Primary	pts/ww/p/ww/sbt/ww
	Secondary	sbt/ww/osr/wb		Secondary	ww/sb/osr
	Set-aside	sbt/ww/sa		Set-aside	p/ww/sa/ww/sbt/ww
Peaty	Primary	sbt/ww/p/ww	Clay	Primary	osr/ww/ww/wbn/ww/ww
	Secondary	wbn/ww/osr/ww		Secondary	osr/ww/ww/wbn/ww/ww
	Set-aside	pts/ww/sa/sbt/ww		Set-aside	osr/ww/ww/sa/ww/wb
Organic	Primary	sbt/ww/p/ww	Shallow	Primary	pts/ww/sb/sbt/ww/wb
	Secondary	wbn/ww/osr/ww		Secondary	sbt/ww/osr/wb
	Set-aside	pts/ww/sa/sbt/ww		Set-aside	sbt/ww/sa
Other Mineral	Primary	osr/ww/ww/wbn/ww/wb	Key: osr = oilseed rape ww = winter wheat wb = winter barley sb = spring barley p = peas sa = set aside pts = potatoes sbt = sugar beet wbn = winter field beans		
	Secondary	osr/ww/ww/sbt/ww/wb			
	Set-aside	osr/ww/ww/sa/ww/wb			

The use of these rotations in the SWAT model did not give a good representation of the crops known to have grown in the study area from EDL data. There was far too much winter wheat being represented in the model due to Wick 2 and Wick 3 soils (which cover the majority of the study area) being classified as 'other mineral' under ADAS standard rotations (Table 5.3.4).

Table 5.3.4: **ADAS Soil Texture Classes**

Predominant Soils	ADAS Texture Class	Area (ha) in SWAT
Altcar	Peaty	621.58
Beccles	Other Mineral	1039.76
Felthrope	Organic	1674.01
Gresham	Other Mineral	1547.74
Hanworth	Organic	2038.95
Isleham	Organic	1463.04
Newport 4	Sandy	3787.67
Wallesea	Clayey	116.06
Wick 2	Other Mineral	31279.82
Wick 3	Other Mineral	17645.70

To better represent EDL data Wick soils were reclassified as 'sandy soils' to help reduce the quantity of winter wheat being grown in the river basin and to increase the area of other crops such as potatoes, spring barley and sugar beet. The Wick soils were chosen as they are usually sandy at depth, and therefore could easily fall into either the 'sandy' or 'other mineral' category.

Rotations were also slightly adjusted to increase or decrease the amount of certain crops within the SWAT model. Set- aside rotations were not used within the model, instead the 5500 ha which were counted as set-aside in the 2000 EDL data was primarily allocated to the wettest soils in the river basin as permanent set-aside, based on the soil wetness class (Table 5.3.5). The system of *wetness class* grades soils from Wetness class I, well drained to Wetness class VI, almost permanently waterlogged within 40cm depth (Hodge *et al.,* 1984).

Table 5.3.5: Soil Wetness Classes (Hodge *et al.*, 1984)

Soil	Wetness Class
Beccles	III
Hanworth	III to V
Wallesea	IV

Maize was not incorporated in the ADAS rotations but is a known crop in the study area, being represented by 282 ha in the EDL data. Maize is recognized as having a higher risk of soil erosion and run-off than most other crops. The late drilling of maize means that the land is not 'covered' with a growing crop until late into June which results in the ground being more vulnerable than winter cereal crops to an intense summer thunderstorm that can lead to flash flooding. With late harvesting it is also necessary for machinery to access the land when it is generally wet and close to field capacity, causing an increased vulnerability to compaction in the soil. Compaction in the soil reduces permeability, increasing the risk or erosion and run-off. As maize is best suited to sandy soils, it was allocated to Newport soil series within the SWAT model.

Therefore, to give a good representation of EDL data within the SWAT model (Fig 5.3.7), 13 rotations have been created based on the ADAS standard rotation information, taking into account the soil type (Table 5.3.6). These were based on averaged data for 3 sets of EDL data covering 1994, 1997 and 2000, as these years fell within the study time period.

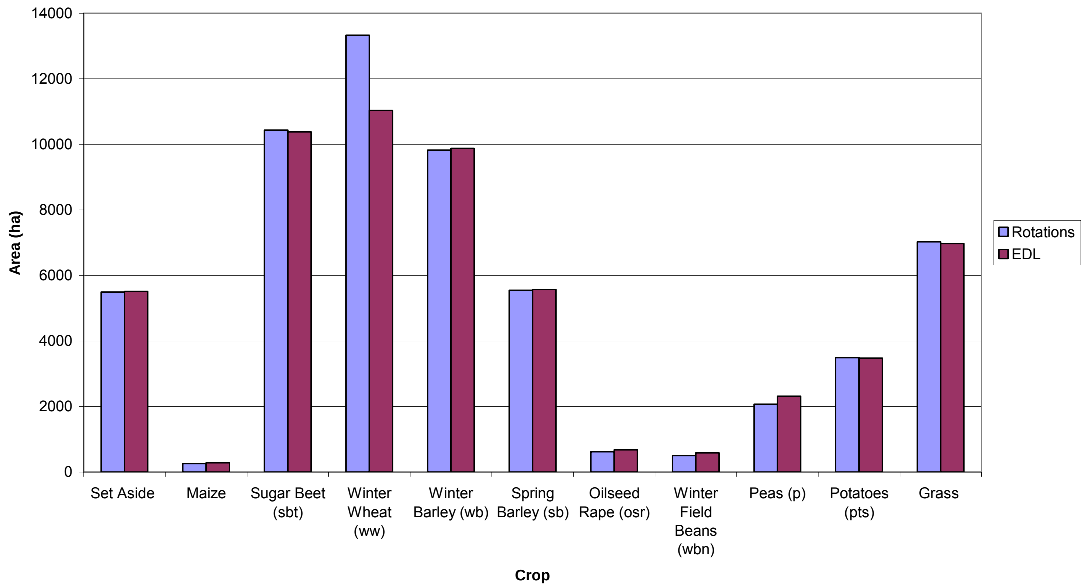


Figure 5.3.7: Comparison of modelled and actual crop areas

Table 5.3.6: Adjusted crop rotations

Other Mineral	Sandy	Organic	Peaty
osr/ww/wb/wbn/ww/wb	pts/ww/sb/sbt/ww/wb	sbt/ww/sb/ww	wbn/ww/p/ww
osr/ww/sbt/wbn/ww/wb	pts/ww/wb/wbn/ww/wb	sbt/ww/p/ww	
	wbn/ww/sb/sbt/ww/wb		
	sbt/ww/sbt/wb		
	sbt/ww/p/wb		
	Maize		
	sbt/ww/osr/wb		
	sbt/ww/sb/wb		

Information on dates of planting along with harvest dates was not available for the study area. Indicative crop calendar dates (Holman et al., 2004) were therefore used in creating the management files for the SWAT model (Table 5.3.7).

Fertiliser Schedule

The fertiliser operation in SWAT applies fertiliser or manure to the soil. Information required in the fertiliser operation includes the timing of the operation (month and day), the type of fertiliser/manure applied, the amount of fertiliser/manure applied and the depth distribution of fertiliser application. SWAT assumes surface run-off interacts with the top 10mm of soil. Nutrients contained in this surface layer are available for transport to the main channel in surface run-off, therefore it is important to obtain the correct data for the fertiliser or manure application operation.

Fertiliser information has been gathered from a number of sources. Application dates have been taken from published reviews (Hough, 1990, Knott *et al*., 1994, Bunting *et al*., 1978, Beukema and Van der Zaag, 1990 and Whitehead, 1995). Fertiliser type and application rates have been taken from best practices guidelines (MAFF, 2000). As well as looking at recommended fertiliser application rates part of the study area falls under Nitrate Vulnerable Zones (NVZ) (Figure 5.3.8). The area coloured in blue shows that the area south of Belaugh within the Bure river basin was designated as a NVZ in 1996; the areas in red were designated as NVZ's in 2002 (DEFRA, 2002a). These areas are designated as NVZ's due to the risk of eutrophication. Under the guidelines for land in NVZ's nitrogen fertiliser application cannot exceed crop requirements and organic manures have a whole farm limit of 210 kg ha^{-1} and 250 kg ha^{-1} for arable and grassland areas respectively. Nitrogen fertilisers and manures cannot be applied on arable land between 1st September and 1st February (DEFRA, 2002a). Table 5.3.7 shows fertiliser application rates and dates for varying crops.

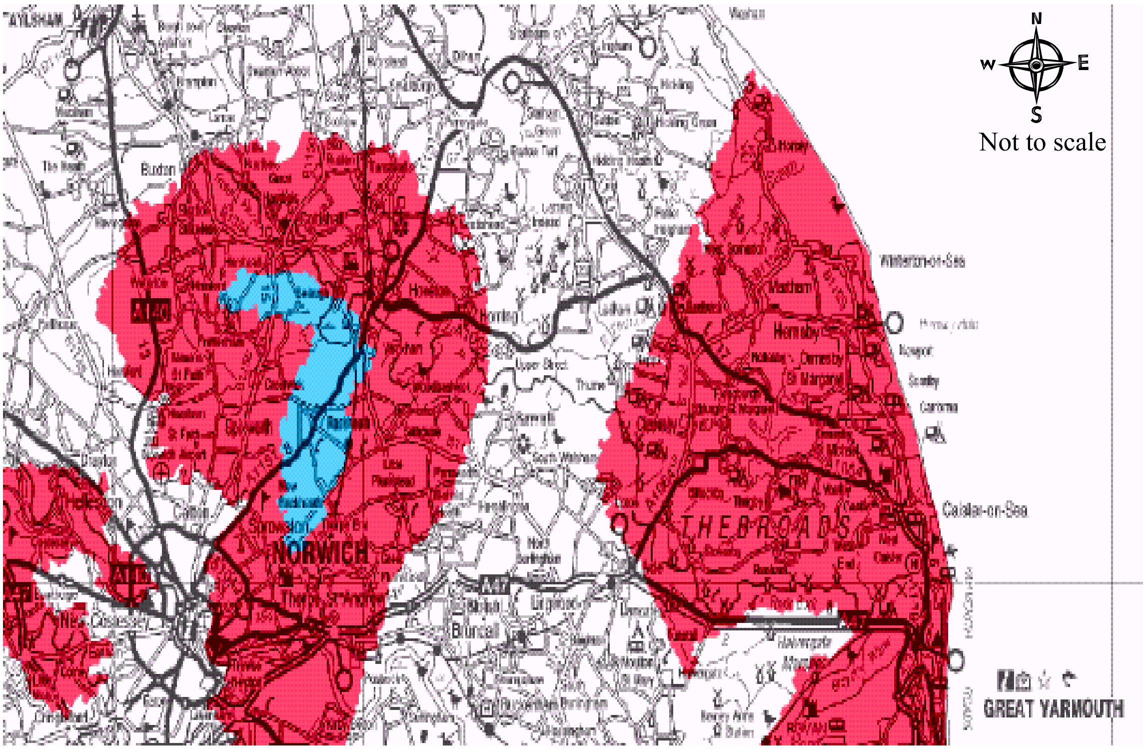


Figure 5.3.8: Nitrate vulnerable zones within the study area (DEFRA, 2002a)

Table 5.3.7: Crop management dates for SWAT (planting, harvest, fertiliser application)

				Soil type and date			
Crop	Operation	Type	Amount (kg ha^{-1})	Other Mineral	Sandy	Organic	Peaty
Oilseed rape	Fertiliser	P	65	Aug-12	Aug-12		
	Tillage	Plough		Aug-25	Aug-25		
	Plant			Aug-31	Aug-31		
	Fertiliser	N 34.5%	40	Feb-23	Feb-23		
	Fertiliser	N 34.5%	80	Mar-22	Mar-22		
	Fertiliser	N 34.5%	80	Apr-23	Apr-23		
	Irrigation			N/A	N/A		
	Harvest/Kill			Jul-30	Jul-30		
Winter wheat	Fertiliser	P	100	Aug-31	Aug-31	Aug-31	Aug-31
	Tillage	Plough		Sep-30	Sep-30	Sep-30	Sep-30
	Plant			Oct-15	Oct-15	Oct-15	Oct-15
	Fertiliser	N 34.5%	40	Mar-13	Mar-13	Mar-13	Mar-13
	Fertiliser	N 34.5%	80	Apr-27	Apr-27	Apr-27	Apr-27
	Fertiliser	N 34.5%	40	May-25	May-25	May-25	May-25
	Irrigation			N/A	May-07	N/A	May-07
	Harvest/Kill			Aug-20	Aug-20	Aug-20	Aug-20
Winter Barley	Fertiliser	P	100	Aug-29	Aug-29		
	Tillage	Plough		Sep-25	Sep-25		
	Plant			Oct-01	Oct-01		
	Fertiliser	N 34.5%	40	Mar-05	Mar-05		
	Fertiliser	N 34.5%	100	Apr-24	Apr-24		
	Irrigation			N/A	N/A		
	Harvest/Kill			Jul-31	Jul-31		
Winter field beans	Fertiliser	K20/P205	50	Sep-28	Sep-28		Sep-28
	Tillage			Oct-20	Oct-20		Oct-20
	Plant			Oct-31	Oct-31		Oct-31
	Irrigation			N/A	N/A		N/A
	Harvest/Kill			Sep-01	Sep-01		Sep-01
Sugar beet	Fertiliser	P	60	Oct-10	Oct-10	Oct-10	
	Tillage	Plough		Oct-20	Oct-20	Oct-20	
	Plant			Mar-31	Mar-31	Mar-31	
	Fertiliser	N 34.5%	40	Apr-11	Apr-11	Apr-11	
	Fertiliser	N 34.5%	30	May-03	May-03	May-03	
	Fertiliser	N 34.5%	30	May-19	May-19	May-19	
	Irrigation			May-30	May-21	May-30	
	Harvest/Kill			Nov-03	Dec-01	Nov-25	
Potatoes	Fertiliser	P	100		Feb-10		
	Tillage	Plough			Feb-19		
	Plant				Apr-01		
	Fertiliser	N 34.5%	75		Apr-01		
	Fertiliser	N 34.5%	75		Apr-18		
	Irrigation				May-04		
	Harvest/Kill				Sep-13		
Spring barley	Fertiliser	P	60		Oct-20	Oct-20	
	Tillage	Plough			Oct-28	Oct-28	
	Plant				Feb-20	Feb-20	
	Fertiliser	N 34.5%	50		Feb-20	Feb-20	
	Tillage	Ring Roll			Feb-29	Feb-29	
	Fertiliser	N 34.5%	50		Apr-05	Apr-05	
	Irrigation				May-05	May-17	
	Harvest/Kill				Aug-08	Aug-08	
Peas	Fertiliser	P	30		Sep-20	Sep-20	Sep-20
	Tillage	Plough			Sep-26	Sep-26	Sep-26
	Plant				Mar-14	Mar-19	Mar-26
	Irrigation				May-12	May-18	May-12
	Harvest/Kill				Aug-01	Aug-06	Aug-13
Maize	Fertiliser	P	45		Nov-01		
	Tillage	Plough			Nov-10		
	Plant				Apr-24		
	Fertiliser	N 34.5%	25		Apr-25		
	Fertiliser	N 34.5%	45		May-26		
	Irrigation				Jun-04		
	Harvest/Kill				Oct-26		
Grass	Irrigation				May-04		

Irrigation Schedule

Irrigation in an HRU can be scheduled by the user or automatically applied by SWAT. In addition to specifying the time and application amount, the user must specify the source of irrigation. For a given irrigation event, SWAT determines the amount of water available in the source. The amount of water available is compared to the amount of water specified in the irrigation operation. If the amount available is less than the amount specified, SWAT will only apply the available water.

Water applied to an HRU is used to fill the soil layers to field capacity beginning with the soil surface layer and working downward until all the water applied is used up or the bottom of the profile is reached. If the amount of water specified in an irrigation operation exceeds the amount needed to fill the soil layers up to field capacity water content, the excess water is returned to the source.

It was hoped that irrigation could be applied automatically by SWAT because inputting irrigation dates and amount for crops such as potatoes could be very time consuming. However, it was not possible to use the auto irrigation, as when attempted, irrigation would only occur within the first year of the crop rotation. Therefore irrigation had to be scheduled manually.

A programme called CropWat (FAO, 1992) has been utilised to produce irrigation schedules for each crop, which can be transferred to the SWAT management files. CropWat requires monthly climate data (temperatures, humidity, wind speed and sunshine), crop files with planting dates, and monthly rainfall data. Data used for SWAT were transferred to CropWat. Climate data were taken from BADC data for Coltishall, as this is the only climate gauge in the river basin. Monthly rainfall was taken from the Aylsham rain gauge as this gauge had data nearest the annual areal average as described in section 5.3.2. From this irrigation was modelled for all years within the study period.

Default soil types were used within the model for light and medium soils. Irrigation scheduling criteria were set to irrigate when 100% of readily available soil moisture depletion occurred. Application depth was set to refill 100% of readily available soil moisture. An example of the model output can be seen in Appendix Eight. Irrigation application dates and net irrigation (mm) were used to schedule irrigation application within the SWAT management files for each crop within the HRU.

It was thought that using CropWat would also yield an advantage over SWAT. When applying irrigation SWAT fills the whole soil profile with water, therefore returning the soil back to field capacity each time water is applied. This does not reflect common UK practice and results in the soil being at field capacity for varying times of the year. For crops such as potatoes irrigation depth should only be between 30 – 50mm depending on the type of soil (FAO, 1989). The hydrology of the system is therefore affected by having the soil at field capacity at the wrong times of the year. However a similar problem occurs within CropWat where the model application depth is set to refill 100% of readily available soil moisture, returning the soil to field capacity. This problem therefore needs to be addressed when using the irrigation function within SWAT in future; and/or a different irrigation model should be used.

5.3.7 Abstraction Data

Water management practices can be one of the most complicated portions of data input for the model because water management affects the hydrologic balance. Within SWAT water abstraction can be modelled through consumptive water use. This is a management tool that removes water from the river basin. Water removed for consumptive use is considered to be lost from the system. SWAT allows water to be removed from the shallow aquifer, the deep aquifer, the reach or the pond within any sub basin in the river basin. Consumptive water use is allowed to vary from month to month. For each month in the year, an average daily volume of water removed from the source can be specified.

Within the Bure and Ant river basin there are 626 licensed abstraction points. These points have been derived from the Environment Agency's database of licensed abstraction points from 1990 – 2003. Abstraction licences provide information on the original effective start date of the licence (not always completed in newer licences), start and end dates of versions of the licence, and lapsed, expired or revoked dates for those licences which are no longer current. These dates have been used to determine when the licences were in existence during the period after 1990. The maximum daily authorised abstraction quantity is supplied in cubic metres, and is the maximum for the licence as a whole. This field is complete for all current licences, but has not necessarily been completed on older revoked licences.

Grid references for abstraction points are given as NGRs. A single abstraction point will have one grid reference, a reach on a watercourse or well point system will have 2 grid references, and an area (water may be taken at any point within an area marked on a map) will have 4 grid references.

ArcView has been used to help allocate each abstraction point to a sub-basin within the SWAT model through the given NGRs. SWAT only allows an average daily volume of water to be removed from each sub-basin. Therefore, the licensed abstraction quantity for each abstraction point within an individual sub-basin has been totalled to give one value per sub-basin (Appendix Nine).

Work carried out by Marcehal (2004) in the East Anglia region (rivers Bure, Wensum and Tud) indicated that the actual amount of water abstracted is approximately only 80% of licensed volumes. As a result of this only 80% of the authorised abstraction quantity has been applied to the SWAT model.

5.3.8 Discharge Data

SWAT directly simulates the loading of water, sediment and other constituents from land areas in the river basin. To simulate the loading of water and other pollutants from sources not associated with land area (e.g. sewage treatment works), SWAT allows point source information to be read in at any point along the channel network.

There are 320 point sources of discharge in the Bure and Ant river basin (Appendix Ten). Information on point sources of discharges were obtained from the EA. Where mean daily flows were not given they were estimated to be 50% of the maximum consented daily flow. Where available, phosphorus inputs were estimated in the same way.

Chapter Six Bure and Ant Model Calibration and Validation

6.0 Model Calibration and Validation

River basin models contain many parameters, some of which cannot be measured. In order to utilise any predictive river basin model for estimating the effectiveness of future potential management practices, the model must be first calibrated to measured data and then should be tested (without further parameter adjustment) against an independent set of measured data. This testing of a model on an independent data set is commonly referred to as model validation. It is standard hydrological modelling practice to divide available time series data into two sets (Klemes, 1986). One set is used for calibration and the remaining data are used for validation.

Model calibration determines the best, or at least a reasonable parameter set, while validation ensures that the calibrated parameter set performs reasonably well under an independent data set. Provided the model can perform well for this independent set of data then it is considered robust and can be used with some confidence for future predictions under different management scenarios.

6.0.1 Calibration and Validation Parameters

Evapotranspiration, infiltration and surface run-off are important components of the water balance for correct representation of nutrient transport and loss. Evapotranspiration is a function of crop growth, therefore only a proper simulation of crop growth will ensure realistic modelling of evapotranspiration and nutrients within a river basin.

Similarly, leaching of nutrients through the soil profile depends on the quantity of water entering and moving through the soil profile in terms of percolation or infiltration. Hence infiltration has to be modelled properly to ensure reasonably accurate simulation of nutrient leaching.

Appropriate modelling of surface run-off is a prerequisite for modelling nutrient run-off. Apart from a match of predicted and observed stream flow and acceptable performance in hydrological modelling, a correct partitioning of water in these three phases is required before nutrient modelling is attempted. With these requirements in mind SWAT is calibrated in three stages:

- Water balance and stream flow
- Sediment
- Nutrients

It is known from previous studies that hydrological models are unlikely to perform well if the calibration period is particularly dry or wet. Data from 1990 – 1994 have been used in the calibration of the Bure and Ant model, as yearly totals were closely matched to annual average rainfall. Data from 1994-1999 will be used in validation of the model. The two periods do overlap but it is common practice to discard the first year of model results as this period is considered as a 'warm up' period for the model. In this period the model is allowed to initialise and then approach reasonable starting values for model state variables and adjust any inaccurate initial conditions. For example, the warm up period allows the model to deposit sediment in the river network and fill the soil partially with soil water before simulation results are considered realistic. The table below shows the available data and periods of data used for the calibration and validation exercise within the Bure and Ant model.

Table 6.0.1: Available data for model calibration and validation

	Calibration			Validation		
Parameter	Site	Dates	Source	Site	Dates	Source
Flow	Honing Lock	1991 - 1994	EA	Honing Lock	1995 - 1999	EA
	Ingworth	1991 - 1994	EA	Ingworth	1995 - 1999	EA
	Horstead Mill	1991 - 1994	EA	Horstead Mill	1995 - 1999	EA
Suspended sediment	Wroxham Rail Bridge	1995 - 1996	Johnes (1996b)	Honing Lock	1999	Johnes (1996b)
	Scarrow Beck	1995 - 1996	Johnes (1996b)			
Nitrate	Honing Lock	1991 - 1994	EA	Honing Lock	1995 - 1999	Johnes (1996b) for 1995-1996, EA for rest
	Scarrow Beck	1991 - 1994	EA	Scarrow Beck	1995 - 1999	Johnes (1996b) for 1995-1996, EA for rest
	Horstead Mill	1991 - 1994	EA	Horstead Mill	1995 - 1999	EA
	Wroxham Rail Bridge	1991 - 1994	EA	Wroxham Rail Bridge	1995 - 1999	Johnes (1996b)for 1995-1996, EA for rest
Total phosphorus	Honing Lock	1991 - 1994	EA	Honing Lock	1995 - 1999	Johnes (1996b) for 1999, EA for rest
	Scarrow Beck	1991 - 1994	EA	Scarrow Beck	1995 - 1996	Johnes (1996b)
	Horstead Mill	1991 - 1994	EA	Horstead Mill	1995 - 1999	EA
	Wroxham Rail Bridge	1991 - 1994	EA	Wroxham Rail Bridge	1995 - 1999	Johnes (1996b) for 1995-1996, EA for rest

6.0.2 Model Performance Statistics

Graphical comparisons of model performance can be made through time series plots of observed and simulated flows and state variables (observed versus simulated values). Time series plots are generally evaluated visually for agreement, or lack thereof, between the simulated and observed values. When observed data are adequate, or uncertainty estimates are available, confidence intervals can then be calculated and so considered in the model performance evaluation. A number of statistical tests can also be considered in model performance evaluation. Model predictive performances relative to available measured data can be evaluated for each constituent by calculating model performance statistics. The four numerical model performance measures used are the coefficient of determination (R^2 coefficient), Nash-Sutcliffe simulation efficiency (E_{NS}) (Nash and Sutcliffe, 1970), daily root mean square (DRMS) and percentage bias (PBIAS).

The R^2 coefficient (unit less) and E_{NS} (unit less) simulation efficiency measure how well the trends in the measured data are reproduced by the simulation results over a specified time period and for a specified time set.

The R^2 coefficient for n time steps is calculated as:

$$R^2 = \frac{[\Sigma^n_{i=1}(\text{simulated}_i - \text{simulated}_{avg})\,(\text{measured}_i - \text{measured}_{avg})]^2}{\Sigma^n_{i=1}(\text{simulated}_i - \text{simulated}_{avg})^2\ \Sigma^n_{i=1}(\text{measured}_i - \text{measured}_{avg})^2}$$

The range of values for R^2 is 1.0 (best) to 0. The R^2 coefficient measures the fraction of the variation in the measured data that is replicated in the simulation model results. A value of 0.0 for R^2 means that none of the variance in the measured data in replicated by the model predictions. On the other hand, a value of 1.0 indicates that all of the variance in the measured data is replicated by the model predictions. Henriksen *et al*. (2003) suggests that a R^2 value > 0.85 is excellent for a hydrological model, values between 0.65 and 0.85 are very good, 0.50 – 0.65 are good, 0.20 – 0.50 are poor and < 0.20 are very poor.

The E_{NS} simulation efficiency for n time steps is calculated as:

$$E_{NS} = 1 - \frac{\Sigma^n_{i=1} (measured_i - simulated_i)^2}{\Sigma^n_{i=1} [measured_i - 1/n\Sigma^n_{i=1} measured_i]^2}$$

E_{NS} values range from 1.0 (best) to negative infinity. E_{NS} is a more stringent test of performance than R^2. E_{NS} measures how well the simulated results predict the measured data relative to simply predicting the quantity of interest by using the average of the measured data over the period of comparison. A value of 0.0 for E_{NS} means that the model predictions are just as accurate as using the measured data average, to predict the measured data. E_{NS} values less than 0.0 indicate the measured data average is a better predictor of the measured data than the model predictions. A value greater than 0.0 indicates the model is a better predictor of the measured data than the measured data average.

DRMS ($m^3 s^{-1}$) simply computes the standard deviation of the model prediction error; a smaller value indicates a better model performance. It is calculated using the following equation:

$$DRMS = \surd\ 1/n\ \Sigma^n_i (q^{sim}_t - (q^{obs}_t)^2)$$

The final statistic PBIAS (%) can be calculated using the following equation.

$$PBIAS = \Sigma^n_i (q^{obs}_t - (q^{sim}_t)^2 / \Sigma^n_i (q^{obs}_t \times 100\%)$$

PBIAS measures the average tendency of the simulated flows to be larger or smaller than their observed counterparts; the optimal value is 0; positive values indicate a model bias toward underestimation.

The model calibration criteria can be further based on recommended error percentages for annual water yields. The Montana Department of Environmental Quality (2005) has generalised information related to model calibration criteria (Table 6.0.2). This criteria has been based upon a number of research papers including; Thomann and Muller, 1982; James and Burges, 1982; Donigian, 1982; ASTM, 1984.

Table 6.0.2: Acceptable model calibration hydrology criteria (Montana Department of Environmental Quality, 2005)

Errors (Simulated-Observed)	Recommended Criteria
Error in total volume	10%
Error in 50% of lowest flows	10%
Error in 10% highest flows	15%
Seasonal volume error (summer)	30%
Seasonal volume error (autumn)	30%
Seasonal volume error (winter)	30%
Seasonal volume error (spring)	30%

6.1 Hydrological Calibration

In addition to simulating measured daily flows, model calibration for hydrology considered base flow predictions. Total river flow is composed of base flow and storm flow. To compare model predicted base flow with measured base flow, estimates of base flow volumes for both simulated and measured flow data are required. A number of methods have been developed to separate base flow from total stream flow (Boorman *et al.,* 1995; Gustard *et al.,* 1980 & Arnold *et al.,* 1995a). Two methods have been used to estimate base flow; these are Base flow Index (BFI) from the soil HOST groups (Boorman *et al.,* 1995) and the Turning Points Method (Gustard *et al.,* 1980). Table 6.1.1 shows the results of base flow separation using the two methods at Horstead Mill, Ingworth and Honing Lock. Figure 6.1.1 shows the separation of base flow from total flow at Ingworth. The turning points method has been used in the calibration to determine whether SWAT was modelling groundwater correctly.

Table 6.1.1: Base flow separation results for the Rivers Bure and Ant

Gauge	BFI (HOST)	IH Turning Points
Ingworth	83%	84%
Horstead Mill	83%	79%
Honing Lock	86%	84%

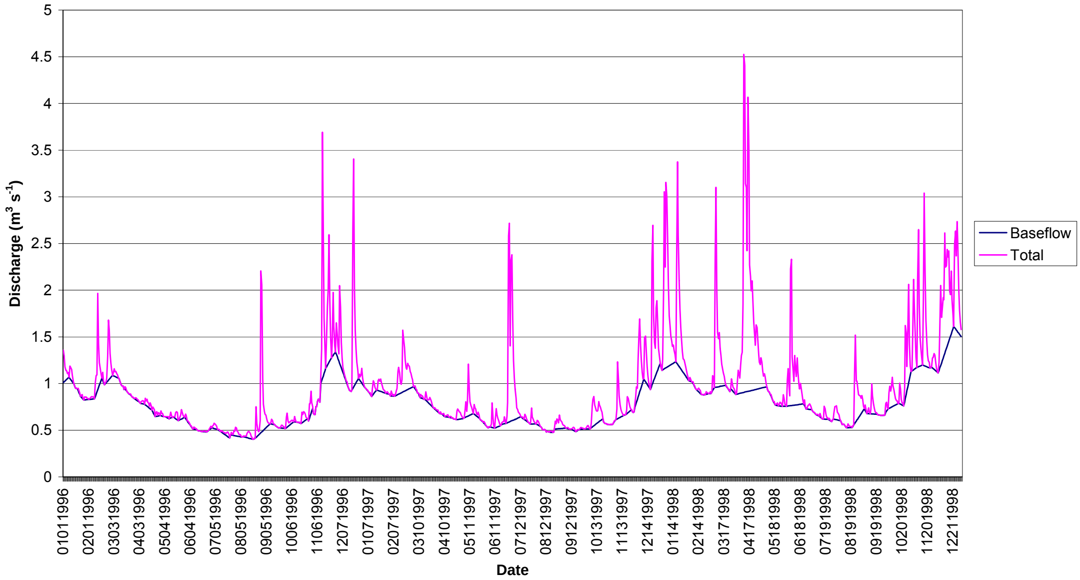


Figure 6.1.1: Base flow separation at Ingworth using the turning points method (1996 – 1998)

There are 8 parameters, which can be adjusted within SWAT when calibrating the water balance within the model (Table 6.1.2). The sensitivity of most of these parameters has been investigated in a number of studies. The curve number, soil water capacity, soil evaporation compensation, saturated hydraulic conductivity and base flow alpha factor have all been evaluated by Arnold *et al.* (2000) in their sensitivity analysis of SWAT. These were also found to be very sensitive in SWAT studies performed by Spruill *et al.* (2000), Santhi *et al.* (2001) and Jha *et al.* (2003a). The groundwater re-evaporation coefficient, minimum depth of water in soil for base flow to occur and minimum depth of water in shallow aquifer for re-evaporation to occur were chosen on the basis of previous SWAT calibration studies (Spruill *et al.,* 2000, Santhi *et al.,* 2001 and Jha *et al.,* 2003a).

Table 6.1.2: Recommend SWAT water balance calibration parameters

Parameter	Notation	Description
Base flow alpha factor	ALPHA_BF	The base flow recession constant is a direct index of groundwater flow response to changes in recharge in days. A change in Alpha will affect the shape of the hydrograph but will not affect the annual water balance.
Available water capacity	AWC	Available water capacity has an inverse relationship with % of rainfall appearing as various water balance components. An increase in AWC value will decrease base flow, surface run-off and hence stream flow
Soil evaporation compensation	ESCO	A change in the value of ESCO will disturb all the water balance components. With an increased ESCO value an increased proportion of rainfall will appear as base flow
Groundwater re-evaporation coefficient	GWREVAP	As GWREVAP approaches 0 movement of water from the shallow aquifer is restricted
Minimum depth of water in soil for base flow to occur	GWQMN	High GWQMN values will result in a considerable portion of rainfall appearing as base flow being retarded. The retarded portion will be stored in the soil
Minimum depth of water in shallow aquifer for re-evaporation to occur	REVAPMN	Movement of water from the shallow aquifer to the root zone or to plants is allowed only if the depth of water in the shallow aquifer is equal to REVAPMN
Saturated hydraulic conductivity	K_{SAT}	An increase in K_{SAT} value will cause an increase in stream flow. The relationship between K_{SAT} and stream flow is direct
Curve Number	CN	Influences the amount of surface run-off calculated for each rainfall event. An increase in CN will increase run-off

6.1.1 Annual Flow Summaries

As no daily time series data are available at the outlet of the river basin being modelled, calibration has been undertaken at three gauged sites within the river basin (Ingworth, Horstead Mill and Honing Lock) Table 6.1.3 shows annual summary statistics for the calibration period, at Ingworth. Exact agreement is unlikely to be reached because of uncertainty involved in estimating which crops are grown from year to year along with soil association distribution and rainfall. This will affect the amount of water required for evapotranspiration and the amount of water left to contribute to stream flow.

Table 6.1.3: Annual SWAT summary statistics for the calibration period at Ingworth (River Bure)

Year	Observed flow (mm)	Predicted flow (mm)	Pred/Obs (mm)	Precipitation (mm)	PET (mm)	ET (mm)
1991	41.63	47.39	1.14	426.10	297.05	129.05
1992	44.72	48.03	1.07	486.60	371.17	115.42
1993	56.73	56.86	1.00	577.20	366.11	211.09
1994	73.92	57.32	0.75	547.70	371.32	176.38

Table 6.1.4 shows that the model is able to accurately predict the overall contributions of groundwater flow to total flow at Ingworth. However, it needs to be remembered that this is only at an annual level.

Table 6.1.4: Observed and predicted groundwater flow contributions to total flow

	Groundwater Flow		Total Stream Flow		% of total stream flow from groundwater	
Year	**Separated (mm)**	**Predicted (mm)**	**Observed (mm)**	**Predicted (mm)**	**Separated**	**Predicted**
1991	37.11	39.78	41.63	47.39	89.14	83.94
1992	38.74	38.70	44.72	48.03	86.63	80.57
1993	47.07	43.47	56.73	56.86	82.97	76.45
1994	62.52	44.72	73.92	57.32	84.58	78.02
Total	**185.44**	**166.67**	**217.00**	**209.60**	**85.83**	**79.75**
					Average Values	

6.1.2 Daily time series

Figures 6.1.2, 6.1.3 and 6.1.4 show the observed and predicted daily flows at Ingworth, Horstead Mill and Honing Lock. The correlation between observed and predicted daily flow is relatively good as can be seen in Table 6.1.5.

Table 6.1.5: Calibrated model performance statistics

			Ingworth		Horstead Mill		Honing Lock	
Statistical test	**Units**	**Optimal values**	**Base flow**	**Total flow**	**Base flow**	**Total flow**	**Base flow**	**Total flow**
R^2	Unit less	1	0.64	0.57	0.7	0.65	0.77	0.58
E_{NS}	Unit less	1	0.57	0.51	0.68	0.64	0.61	0.54
DRMS	$m^3 s^{-1}$	Smaller value indicates a better model performance	0.21	0.27	0.39	0.42	0.04	0.08
PBIAS	%	0 (+ values = bias to underestimate & - values = bias to overestimate)	-10.11	-5.68	4.42	3.49	10.39	4.87

It has been shown that groundwater flows contribute approximately 80% of total water at all sites. It can be concluded that the overall prediction of flow pattern is acceptable, although summer low flows are over predicted, especially at Ingworth on the River Bure, which is represented by the negative PBIAS value both for base and total flow figures. The positive PBIAS figures for Horstead Mill and Honing Lock is a result of the under estimation of peak flows within the model as can be seen in Figures 6.1.3 and 6.1.4. R^2 values fall into the 'good' and 'very good' categories as defined by Henriksen *et al*. (2003).

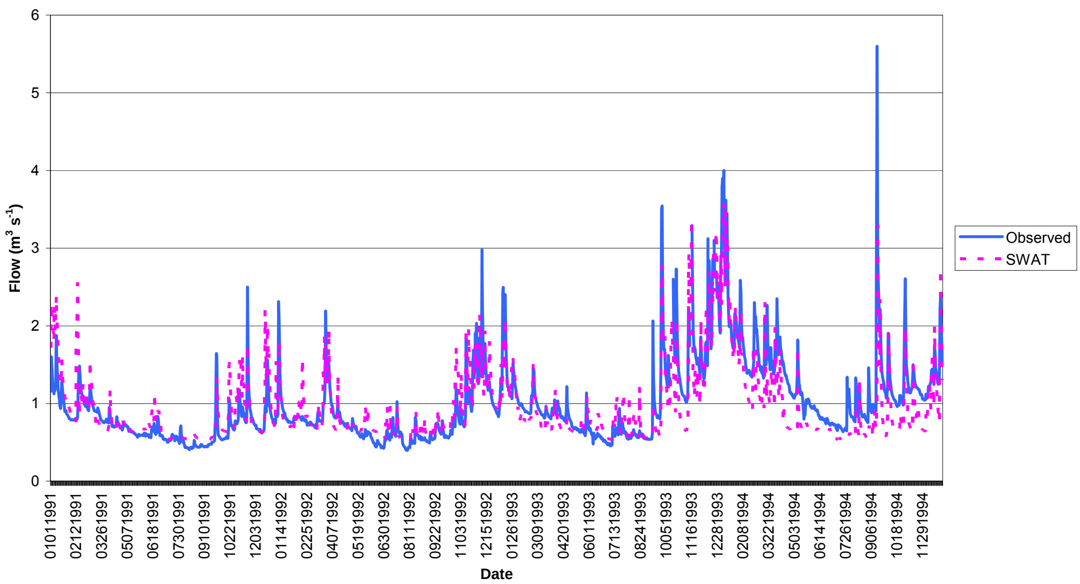


Figure 6.1.2: Calibrated predicted and observed flow comparison at Ingworth (River Bure 1991-1994)

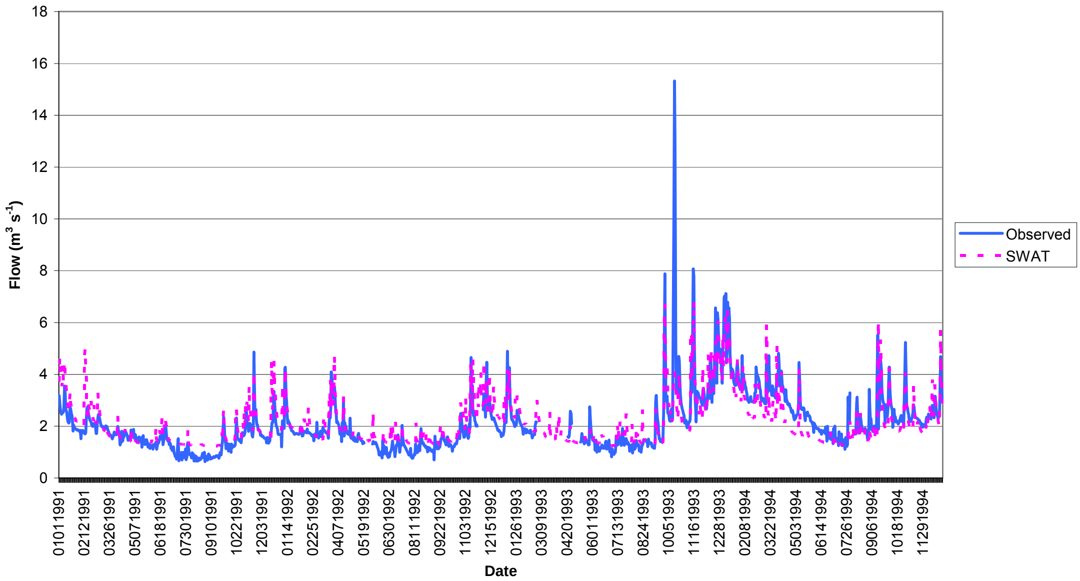


Figure 6.1.3: Calibrated predicted and observed flow comparison at Horstead Mill (River Bure 1991 – 1994)

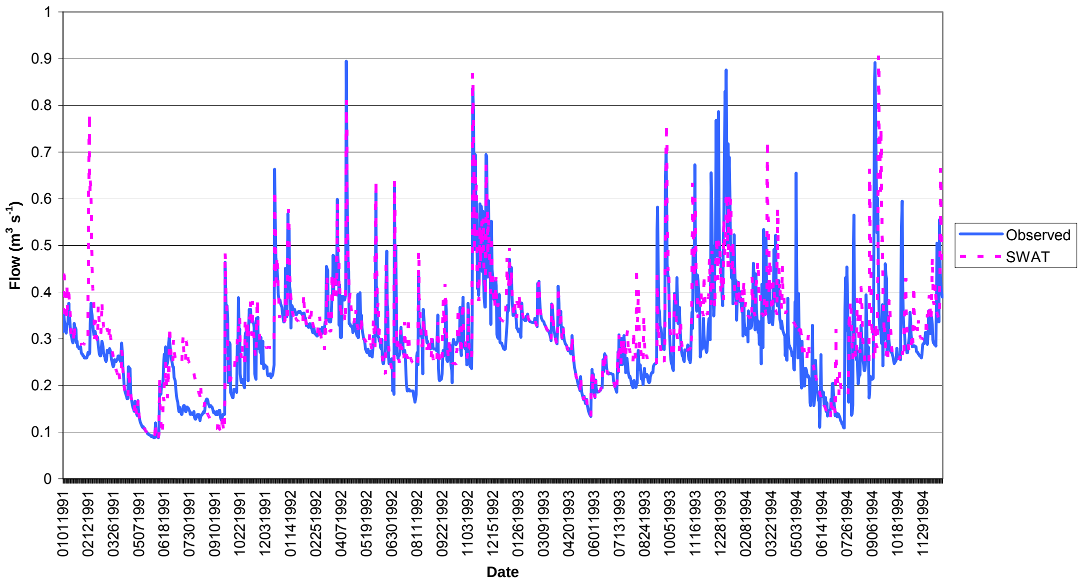


Figure 6.1.4: Calibrated predicted and observed flow comparison at Honing Lock (River Ant 1991 – 1994)

Table 6.1.6 shows the acceptable hydrological calibration criteria in terms of percentage errors for the three calibration sites as recommended in various published texts. The seasons have been defined as:

Summer	June, July, August
Autumn	September, October, November
Winter	December, January, February
Spring	March, April, May

Table 6.1.6: Acceptable hydrological calibration criteria in terms of percentage errors

Errors (Simulated-Observed)	Recommended Criteria	Ingworth	Horstead Mill	Honing Lock
Error in total volume	10%	3.87%	6.72%	4.64%
Error in 50% lowest flows	10%	1.00%	5.90%	7.78%
Error in 10% highest flows	15%	2.30%	6.70%	1.66%
Seasonal volume error (summer)	30%	3.29%	13.21%	9.72%
Seasonal volume error (autumn)	30%	6.79%	0.53%	2.29%
Seasonal volume error (winter)	30%	1.01%	5.76%	2.21%
Seasonal volume error (spring)	30%	15.18%	8.28%	3.88%

It can be seen from the above table that for all the calibration locations with the Bure and Ant SWAT model the percentage errors are low. The sites all fall well below the hydrological calibration criteria recommended in various published texts, with the highest percentage error occurring in spring and summer total flow values. This is also mirrored in the higher percentage errors for the 50% lowest flows.

6.2 Model Performance Indicators

When looking at river flows it is important to look at other model performance indicators before calibrating sediment loading and water quality parameters. Crop growth, evapotranspiration and soil moisture data can all be extracted from the SWAT results files. All of these can affect the water balance within the model, for example soil moisture will affect surface run-off.

Figure 6.2.1 gives examples of crop growth profiles for four dominant crops in the Bure and Ant watershed: winter wheat, winter barley, spring barley and sugar beet, covering 13642 ha, 9821 ha, 5548 ha and 10435 ha respectively. The two lines show the development of leaf-area index (LAI) (in blue) and plant biomass (in pink) during the growing season. The graphs have been used to check that the plants are developing correctly in the model. After planting, dormancy occurs before the crops begin to grow without being subjected to any stresses caused by a lack of nutrients. Plants may also be stressed in SWAT by lack of water or by temperatures outside of the optimal growing range. Published peak LAI's are reported by Hough (1990) to lie between 3-8 for all the above crops. All crops with the exception of sugar beet fall into the lower end of this range. All autumn planted crops show a period of dormancy. Leaf area index (LAI) decreases once plants reach senescence, and biomass falls somewhat later at harvest. It can be seen that all plants are growing in the expected manner.

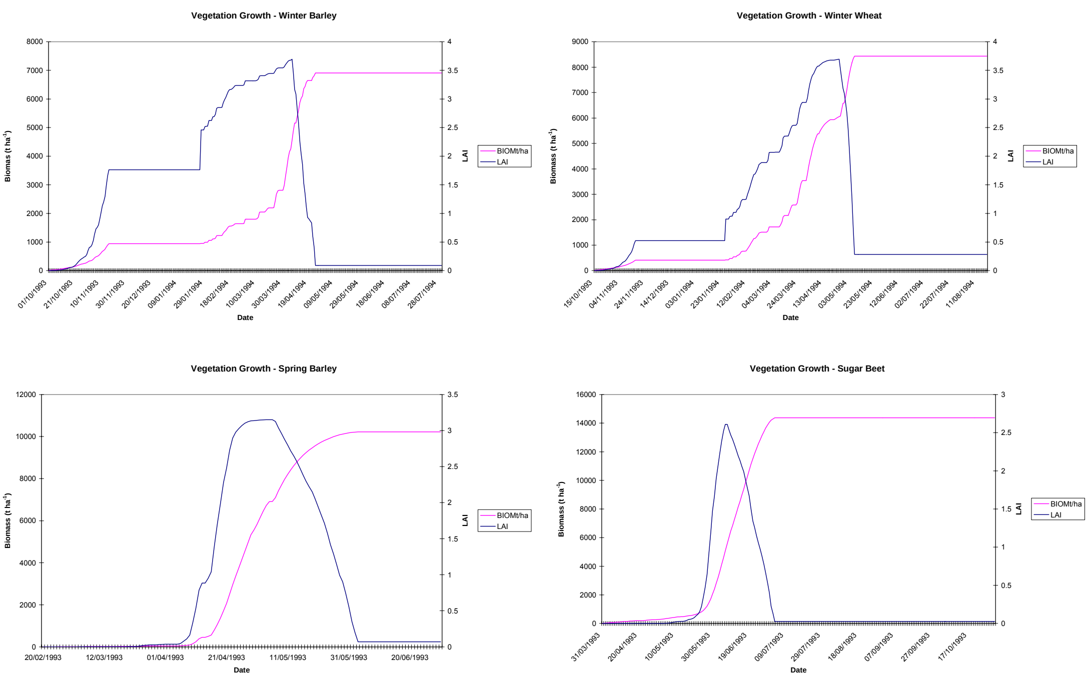


Figure 6.2.1: Modelled plant growth for major crops in the study area

The correct modelling of plant growth is essential to achieve accurate evapotranspiration, as evapotranspiration rate is strongly influenced by a number of vegetative surface characteristics (Penman, 1956). Evapotranspiration is the primary mechanism by which water is removed from a watershed. Roughly 62% of the precipitation that falls on the continents is evapotranspired. Evapotranspiration exceeds run-off in most river basins and on all continents except Antarctica (Dingman, 1994). The difference between precipitation and evapotranspiration is the water available for human use and management. An accurate estimation of evapotranspiration is critical in the assessment of water resources and the impact of climate and land use change on those resources. Figure 6.2.2 shows SWAT predicted potential evapotranspiration values compared to two published sources. A good match can be seen between SWAT and the MORECS data. SWAT predictions are higher then those provided by Smith (1976). This is to be expected due to changes in climate since the publishing of this data in the 1970's.

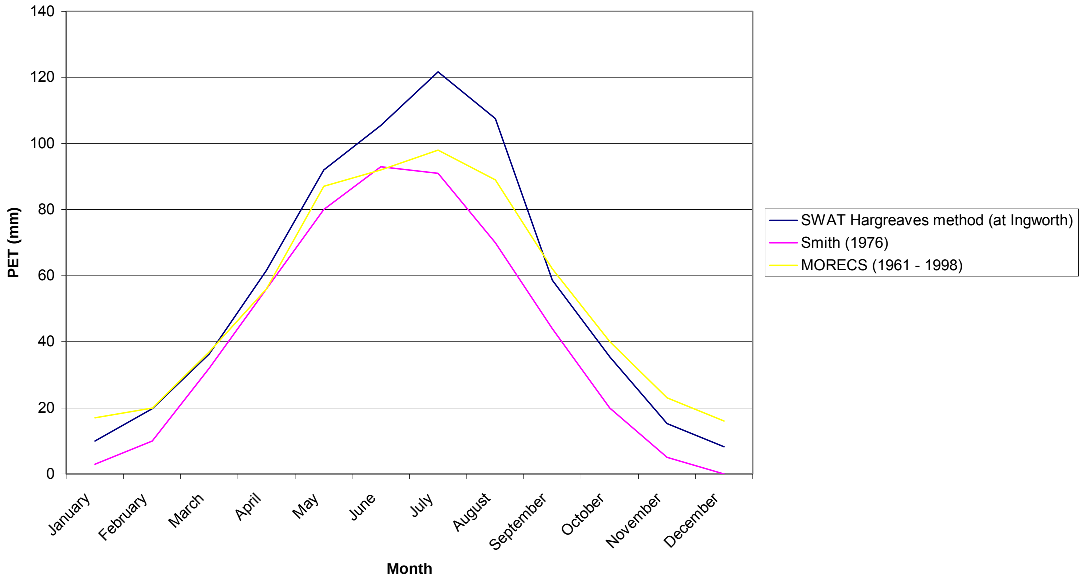


Figure 6.2.2: Average annual predicted PET comparison at Ingworth (River Bure)

If crops are growing correctly and soils have been correctly parameterised then a correct representation of soil moisture development during the year is expected. Figure 6.2.3 shows modelled soil moisture content for the Hanworth soil series in sub-basin 29 of the SWAT model (graphs for other soil series can be seen in Appendix Eleven).

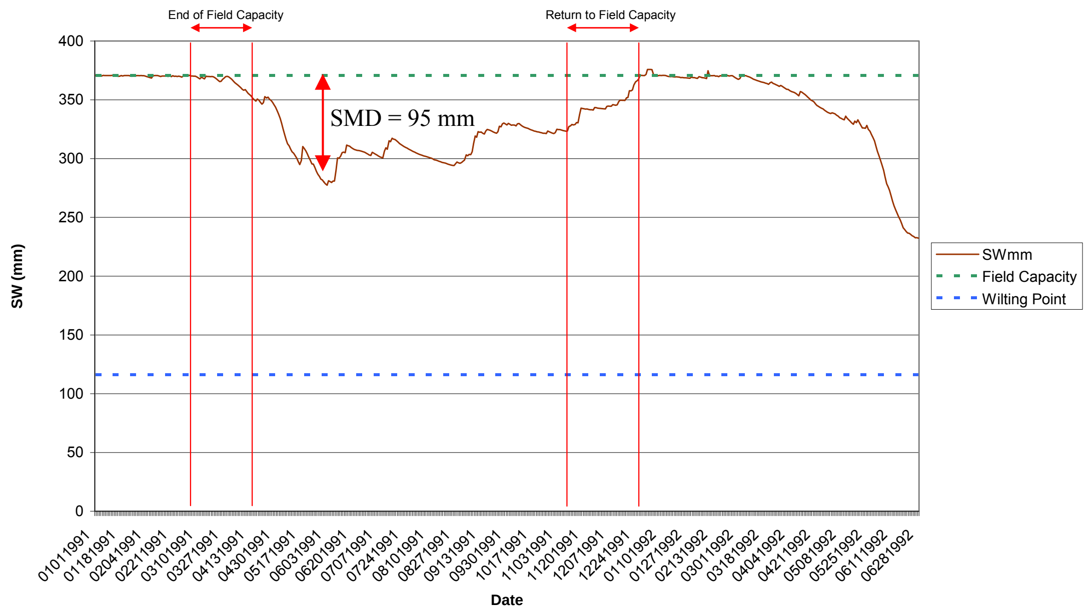


Figure 6.2.3: Comparison between predicted and published soil water content for the Hanworth soil series (1991 – 1992)

Figure 6.2.3 shows the permanent wilting point of the soil (blue line). At this point the plants growing in it will wilt and not recover. This point is however never reached in any of the soils used within the SWAT model. Therefore, none of the crops growing will die due to water stress.

The field capacity of the soil is also shown within Figure 6.2.3 (green line). This is the water present in a soil that has been saturated and allowed to drain until all drainage has ceased. Smith (1976) suggests that in the study area the soil should end field capacity around the middle of March to the end of April (first time period shown in red in Fig. 6.2.3) and return to field capacity (when rainfall exceeds transpiration and the soil gradually re-wets) between December and January (second time period shown in red in Fig. 6.2.3). SWAT is predicting return to field capacity towards the end of the published time period. This may be due to drier summers than those reported by Smith (1976). This late return to field capacity should not greatly affect soil water recharge as spring and autumn are the times for greatest recharge; however it is affecting crops planted in the winter. A period where there is no increase in either crop LAI or biomass can be seen for winter wheat and barley (Figure 6.2.1). This does not affect peak LAI values as discussed previously and field capacity is also being modelled reasonably correctly within SWAT.

In the summer there is not enough rainfall to meet the plant's demand for water and a soil water deficit develops. This is the dryness of the soil in terms of water required to return the soil to field capacity. In the study area it should be between 95 – 123mm (Smith, 1976). The model in this case is showing 95mm (Fig. 6.2.3). It can be concluded that the model is predicting soil moisture very well.

Although SWAT is predicting soil moisture well the problems with the depth of irrigation as discussed in Chapter Five is apparent in the soil moisture outputs from SWAT highlighted in Figure 6.2.4. It can be seen from the graph during the year the soil moisture content increases sharply, returning to near field capacity before the correct date. This has little impact on the overall soil moisture content as soil moisture is only at field capacity for short periods of time and does not affect either LAI or plant biomass as can be seen in Figure 6.2.1. All other parameters such as evapotranspiration, base flow and total river flows are also behaving in the expected way as discussed previously in this chapter.

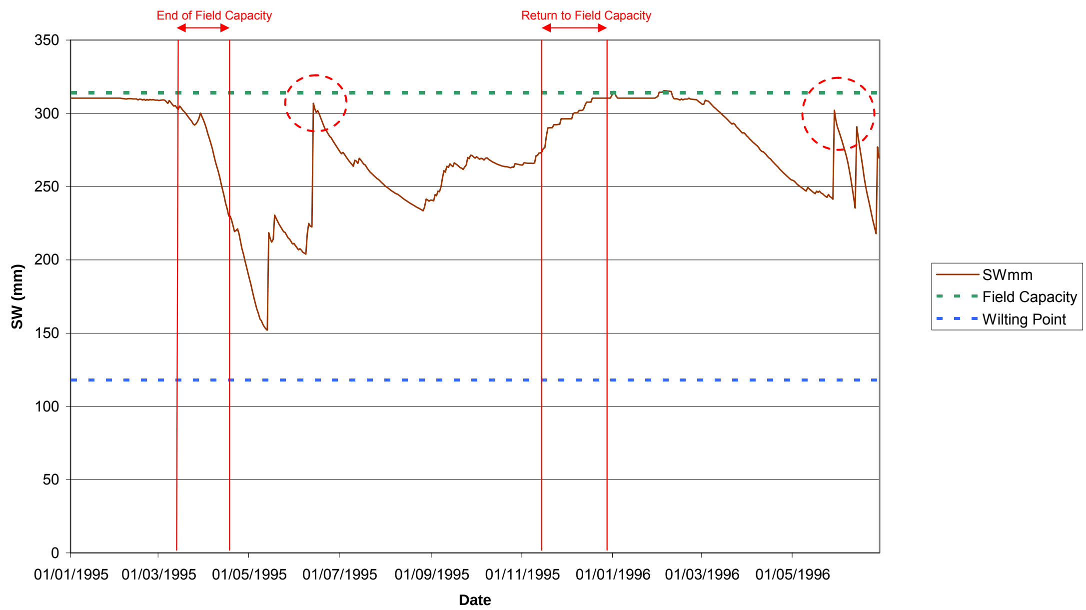


Figure 6.2.4: The effect of irrigation depth on soil water content within SWAT for Wick 3 soil series (1995 – 1996)

6.3 Sediment Calibration

Correct representation of sediment yield is required as one of the major mechanisms for loss of phosphorus from fields to rivers as material sorbed to sediment particles. Once the ratio of surface run-off to base flow contribution to stream flow has been calibrated, the sediment contribution (loadings from HRUs/sub basins) can be calibrated. There are two sources of sediment in SWAT: loadings from HRUs/sub basins and channel degradation/deposition. There are no data to assess the channel degradation/deposition loadings from sub basins. Calibration has been undertaken using 6 parameters (Table 6.3.1). It has then been assumed that the remaining difference between actual and observed is due to channel degradation/deposition.

Table 6.3.1: Recommended SWAT sediment calibration parameters

Parameter	**Notation**	**Description**
USLE crop management factor	USLE_P	The support practice factor is defined as the ratio of soil loss with a specific support practice to the corresponding loss with up-and-down slope culture. Support practices include contour tillage, strip cropping on the contour, and terrace systems. An increase in USLE_P will increase soil loss.
USLE slope length	SLSUBBSN	This is the USLE topographic factor. An increase in slope length will increase sediment yield.
Slope of HRU's	SLOPE	ArcView measures the average slope steepness. This can be increased or decreased by the user to either increase or decrease sediment yield.
Crop practice factor	USLE_C	Value of USLE_C factor for water erosion applicable to the land cover/plant. A decrease in USLE_C to account for local conditions will decrease sediment yield.
Crop residue factor	RSDCO	The plant residue decomposition coefficient is the fraction of residue that will decompose in a day assuming optimal moisture, temperature, C:N ratio, and C:P ratio. If appropriate for the plant a decrease in RSDCO will reduce sediment yield.
Bio-mixing efficiency	BIOMIX	Biological mixing is the redistribution of soil constituents as a result of the activity of biota in the soil (e.g. earthworms, etc.). A decrease in BIOMIX will decrease sediment yield.

No EA data existed for the study period. Observed data from Johnes (1996b) was used for calibration and validation purposes. The lack of available observed sediment data meant that calibration could only be undertaken for a one year period at two locations (Scarrow Beck: Figure 6.3.2 and Wroxham Rail Bridge: Figure 6.3.3). The calibrated parameters were then applied to the remaining sub-basins within the SWAT model. A third sub-basin (Honing Lock) was then used to validate the calibrated sediment parameters.

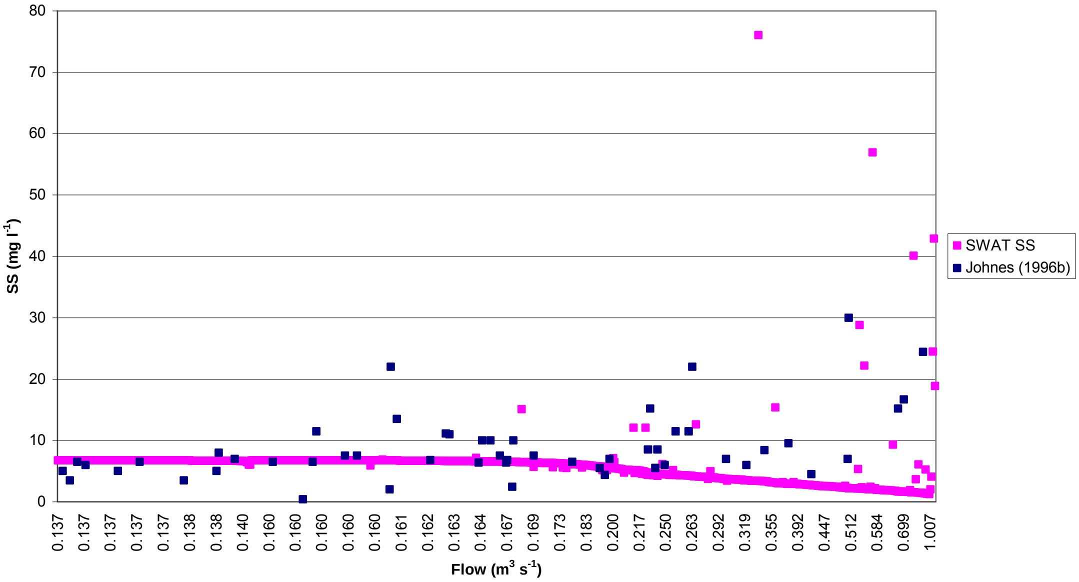


Figure 6.3.2: Sediment calibration at Scarrow Beck (1995 -1996)

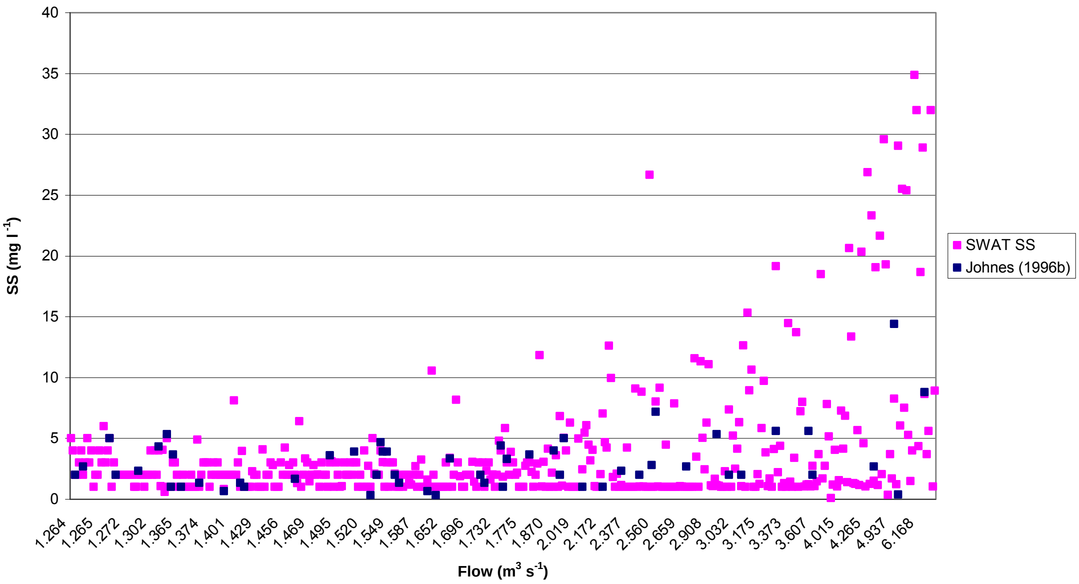


Figure 6.3.3: Sediment calibration at Wroxham Rail Bridge (1995 – 1996)

6.4 Nutrient Calibration

The nutrients of concern in SWAT are nitrate, soluble phosphorus, organic nitrogen and organic phosphorus. For calibration purposes comparisons can be made between EA nitrate measurements and SWAT predicted nitrate measurements. There are two general forms of phosphorus in the main model output files. These are called mineral phosphorus and organic phosphorus in SWAT model documentation and output. Therefore SWAT simulated total phosphorus (mineral and organic phosphorus) and monitored EA total phosphorus are assumed equivalent for calibration. Table 6.4.1 shows SWAT calibration parameters for nutrients.

Table 6.4.1: Recommend SWAT nutrient calibration parameters

Parameter	Notation	Description
Concentration of nitrogen and organic nitrogen in soils	SOL_NO3 and SOL_ORGN	Initial NO_3 and organic nitrogen concentration in the soil layer. This can increase or decreased to realistic levels.
Fertilizer application rates	FRT_LY1	Fraction of fertilizer applied to top 10mm of soil. The remaining fraction is applied to the 1st soil layer below 10mm. An increase in fertilizer application rate will increase nutrient loading.
Crop residue coefficient	RSDCO	The plant residue decomposition coefficient is the fraction of residue that will decompose in a day assuming optimal moisture, temperature, C:N ratio, and C:P ratio. If appropriate for the plant an increase in RSDCO can increase nutrient loading.
Bio-mixing efficiency	BIOMIX	Biological mixing is the redistribution of soil constituents as a result of the activity of biota in the soil (e.g. earthworms, etc.). A decrease in BIOMIX will increase nutrient load.
Nitrogen percolation coefficient	NPERCO	NPERCO controls the amount of nitrate removed from the surface layer in runoff relative to the amount removed via percolation.
Concentration of soluble and organic phosphorus in soils	SOL_MINP and SOL_ORGP	Initial mineral and organic phosphorus concentration in the soil layer. This can be increased or decreased to realistic levels.
Phosphorus percolation coefficient	PPERCO	The phosphorus percolation coefficient is the ratio of the soluble phosphorus concentration in the surface 10 mm of soil to the concentration of phosphorus in percolate. An increase in PPERCO will increase phosphorus loading.
Phosphorus soil partitioning coefficient	PHOSKD	The phosphorus soil partitioning coefficient is the ratio of the soluble phosphorus concentration in the surface 10 mm of soil to the concentration of soluble phosphorus in surface runoff. An increase in PHOSKD will increase soluble phosphorus loading.

Nutrient calibration was undertaken for four sites using EA monthly data as shown in Figures 6.4.1, 6.4.2, 6.4.3 and 6.4.4.

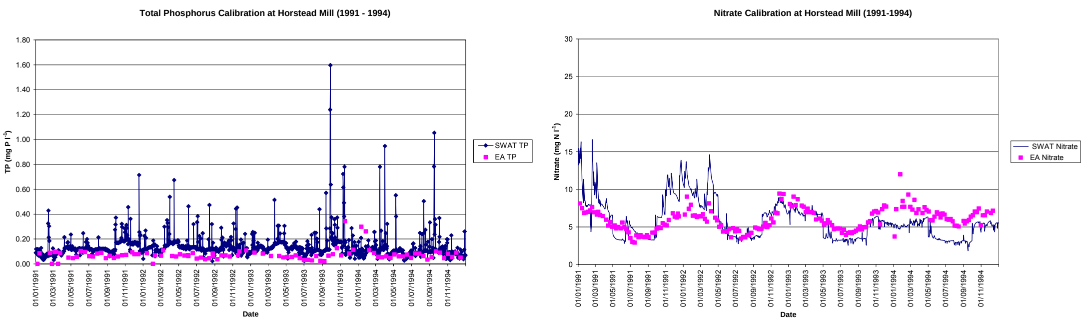


Figure 6.4.1: Nutrient calibration at Horstead Mill

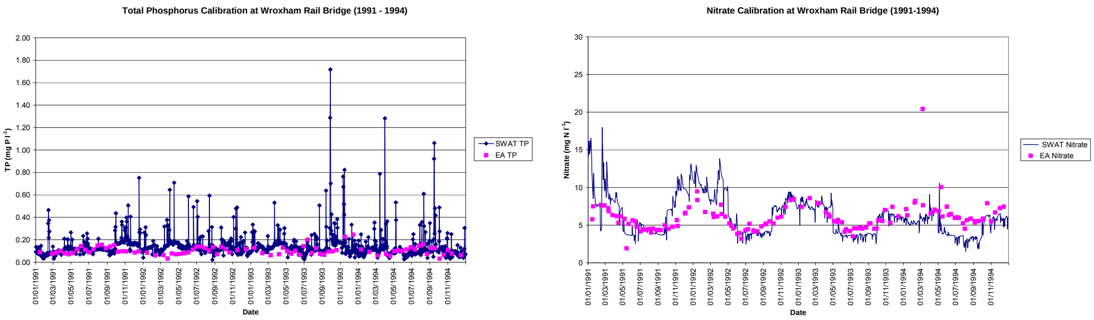


Figure 6.4.2: Nutrient calibration at Wroxham Rail Bridge

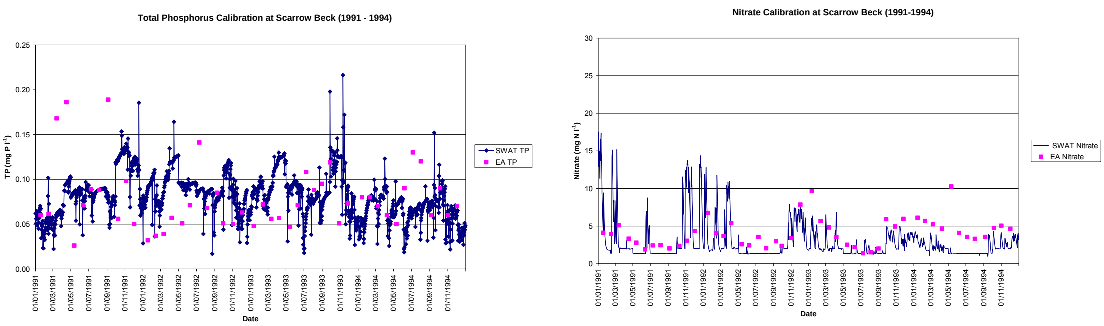


Figure 6.4.3: Nutrient calibration at Scarrow Beck

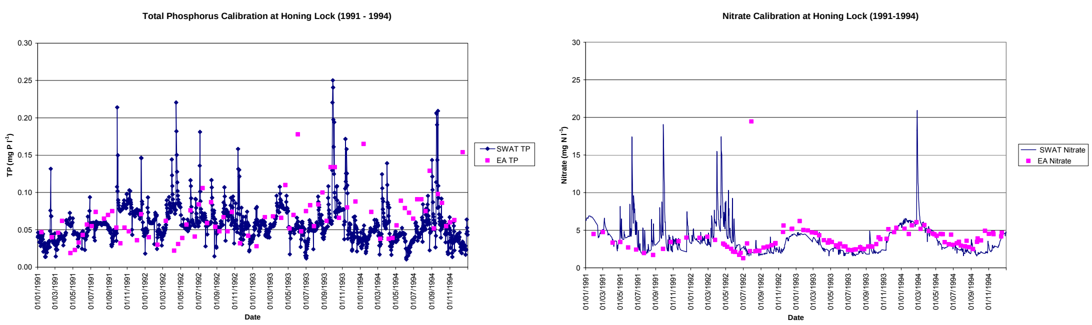


Figure 6.4.4: Nutrient calibration at Honing Lock

Table 6.4.2 shows the performance statistics for nutrient calibration. DRMS values for TP are very low at all the calibration sites, suggesting the model is performing well for TP, but have a tendency to overestimate TP concentrations, reflected in the negative PBIAS values. PBIAS values for N are positive and therefore the model is underestimating N concentrations at all the calibration sites. Overall the model is predicting nutrient concentrations well, with both nutrient parameters falling into the 'good' category (0.50 – 0.65) for R^2 values under Henriksen *et al*. (2003) categories of goodness of fit for a given model.

Table 6.4.2: Performance statistics for nutrient calibration

			Horstead Mill		**Wroxham**		**Scarrow Beck**		**Honing Lock**	
Statistical test	**Units**	**Optimal values**	**N**	**TP**	**N**	**TP**	**N**	**TP**	**N**	**TP**
R^2	Unit less	1	0.51	0.56	0.51	0.58	0.50	0.52	0.50	0.55
E_{NS}	Unit less	1	0.67	0.65	0.67	0.62	0.52	0.59	0.53	0.71
DRMS	mg l^{-1}	Smaller value indicates a better model performance	2.30	0.08	2.43	0.13	2.96	0.05	2.12	0.04
PBIAS	%	0 (+ values = bias to underestimate & - values = bias to overestimate)	2.41	-3.13	3.87	-7.67	9.01	-9.52	3.59	2.80

6.5 Model Validation

Data from 1994 – 1999 have been used to validate the Bure and Ant SWAT model. The results for flow validation can be seen in the table and figures below. Under the Henriksen *et al*. (2003) categories of goodness of fit for a given model the R^2 values for Ingworth, Horstead Mill and Honing Lock base flow fall into the 'good' category (0.50 – 0.65). The total flow at Ingworth and Horstead Mill are in the 'very good' category (0.65 – 0.85), where as total flow at Honing Lock falls in the 'poor' category (0.20 – 0.50).

Table 6.5.1: Validation model performance statistics

			Ingworth		Horstead Mill		Honing Lock	
Statistical test	**Units**	**Optimal values**	**Base flow**	**Total flow**	**Base flow**	**Total flow**	**Base flow**	**Total flow**
R^2	Unit less	1	0.61	0.72	0.61	0.73	0.61	0.43
E_{NS}	Unit less	1	0.47	0.68	0.57	0.69	0.57	0.37
DRMS	$m^3 s^{-1}$	Smaller value indicates a better model performance	0.21	0.32	0.38	0.58	0.05	0.09
PBIAS	%	0 (+ values = bias to underestimate & - values = bias to overestimate)	-2.85	-2.86	-3.34	-4.92	2.53	6.79

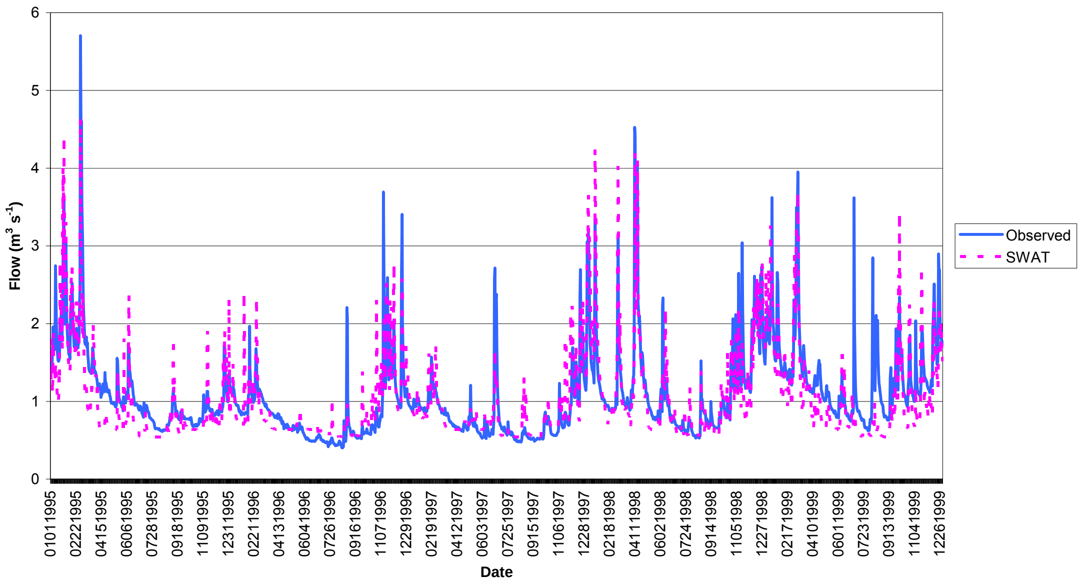


Figure 6.5.1: Flow validation at Ingworth (1995 – 1999)

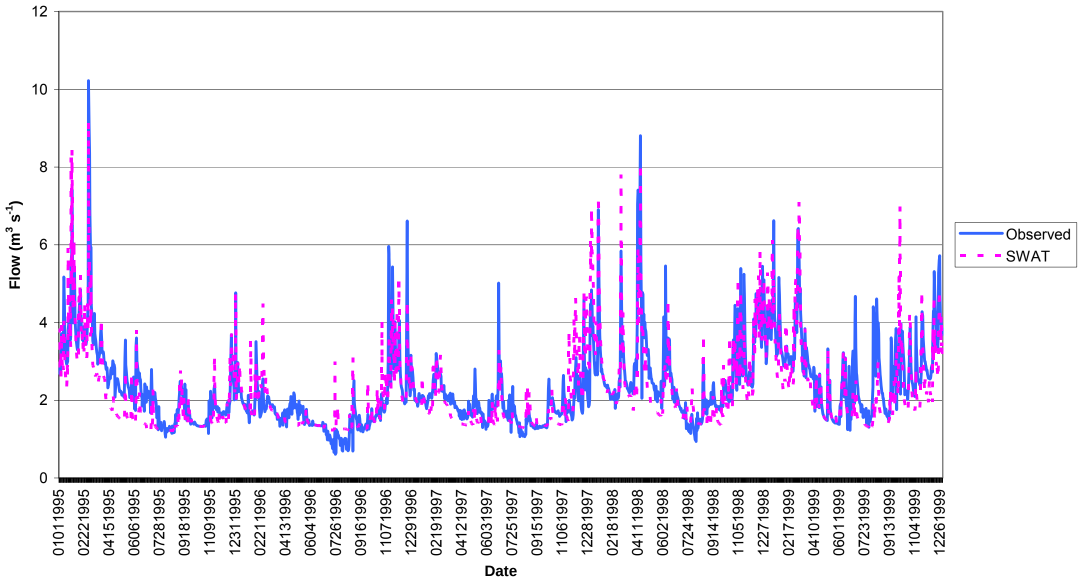


Figure 6.5.2: Flow validation at Horstead Mill (1995 – 1996)

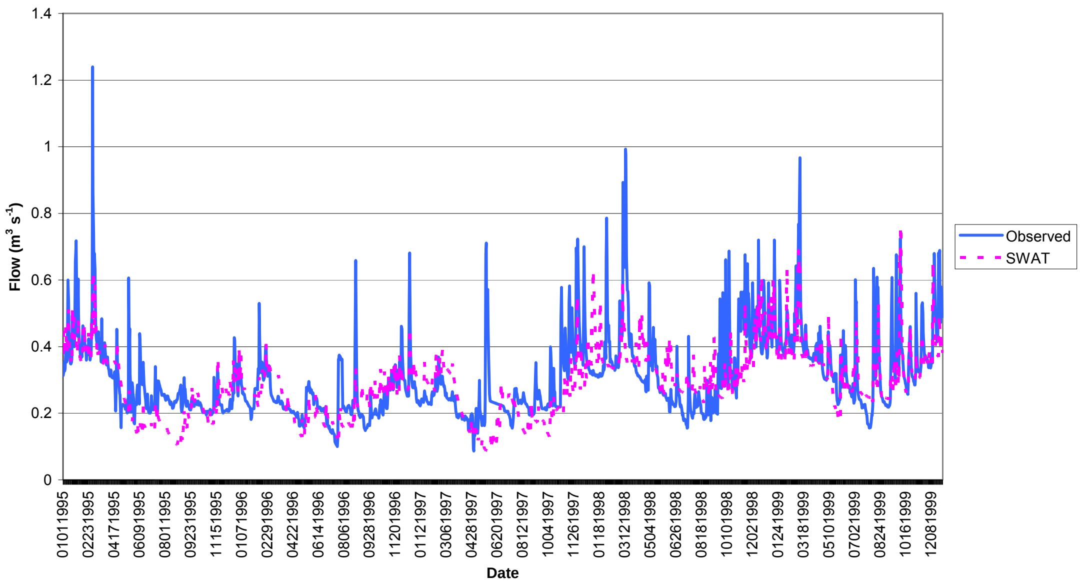


Figure 6.5.3: Flow validation at Honing Lock (1995 – 1999)

Sediment validation was only undertaken at one site: Honing Lock and only for a one year period due to lack of observed data. Figure 6.5.2 shows sediment validation results. Predicted data were compared against observed data provided by Johnes (1996b). It can be seen from the graph below, sediment validation at Honing Lock is on the whole poor. This is a consequence of the poor hydrological validation at this site, only achieving an E_{NS} of 0.37 for total flow during the validation period and a positive PBIAS of 6.79 due to the underestimation of peak flows by SWAT. The under prediction of peak flows at Honing Lock can be accounted for by the poor rainfall data set used in this section of the model. A rain gauge at South Repps (located at the top of the River Ant basin) was used, as this was the gauge, which fell within the River Ant basin. Unfortunately a number of day's data were missing and were therefore infilled using the BADC programme. This is the only site where daily sediment data are available; unfortunately it is also the only set of suspended sediment data, which falls within the validation period. Validation at other sites to check the calibration parameters is not possible.

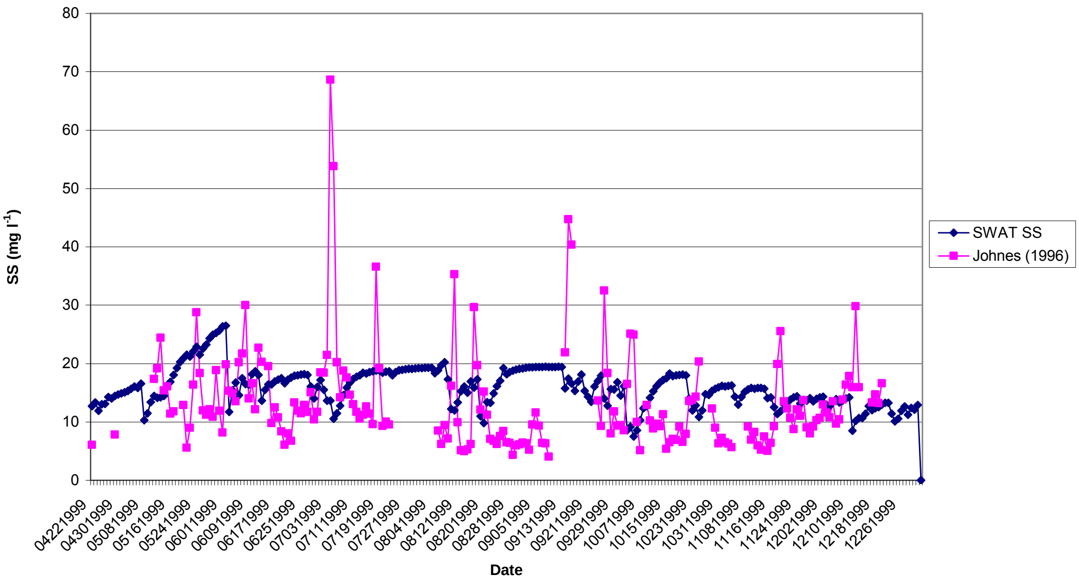


Figure 6.5.4: Sediment validation at Honing Lock (1999)

The validation of nutrients was undertaken at the same four sites as for calibration, using both EA monthly data and Johnes (1996b) weekly values. Figures 6.5.5, 6.5.6, and 6.5.7 and 6.5.8 show validation results for all sites.

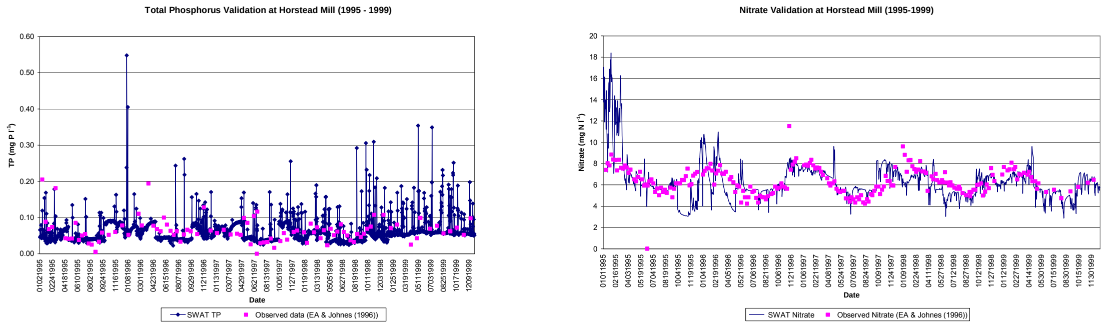


Figure 6.5.5: Nutrient validation at Horstead Mill

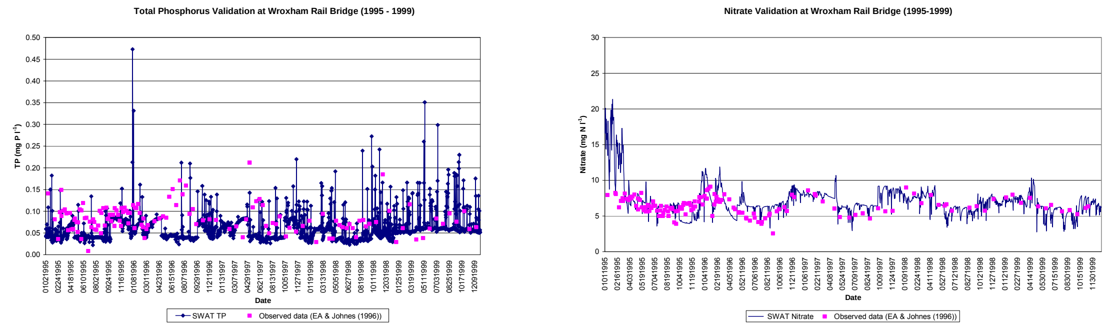


Figure 6.5.6: Nutrient validation at Wroxham Rail Bridge

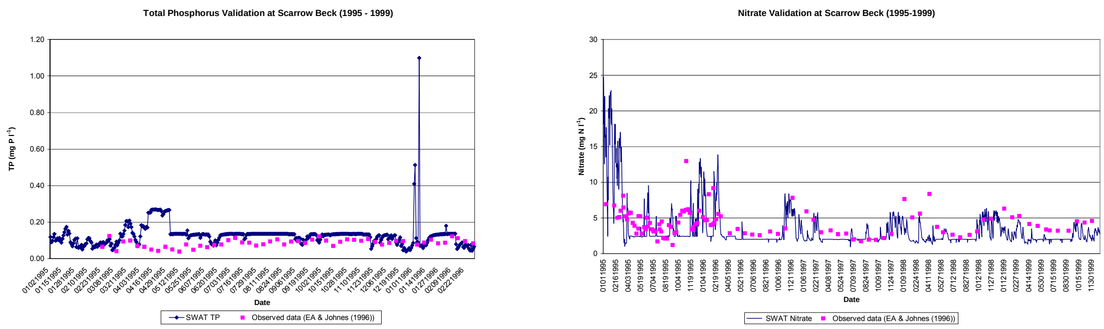


Figure 6.5.7: Nutrient validation at Scarrow Beck

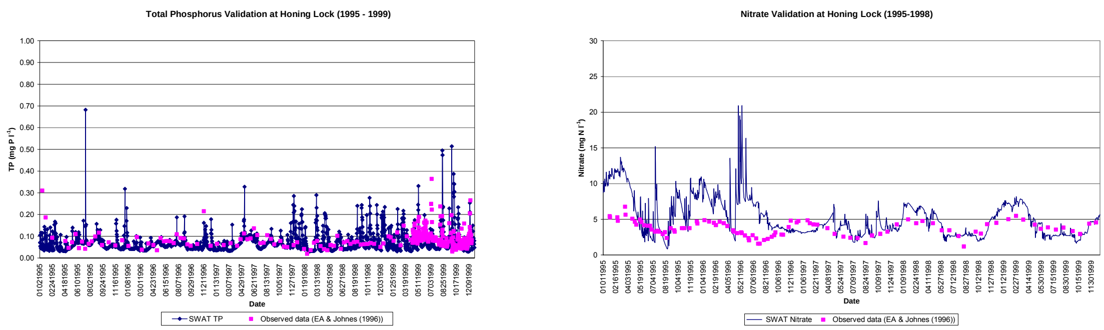


Figure 6.5.8: Nutrient validation at Honing Lock

Table 6.5.2 shows the performance statistics for nutrient calibration. Like the calibration results the DRMS values for TP are very low at all the calibration sites, suggesting the model is performing well for TP, but has a tendency to overestimate TP concentrations, reflected in the negative PBIAS values. PBIAS values for N are positive and therefore the model is underestimating N concentrations at all the calibration sites. Both R^2 and E_{NS} values have slightly improved in the validation period this corresponds with the general improvement seen in the hydrological validation performance statistics. At Honing Lock TP values for R^2 and ENS have decreased from 0.55 and 0.71 to 0.52 and 0.54 respectively. These values reflect the poor validation for total flow at Honing Lock.

Table 6.5.2: Performance statistics for nutrient validation

			Horstead Mill		**Wroxham**		**Scarrow Beck**		**Honing Lock**	
Statistical test	**Units**	**Optimal values**	**N**	**TP**	**N**	**TP**	**N**	**TP**	**N**	**TP**
R^2	Unit less	1	0.53	0.58	0.51	0.58	0.54	0.56	0.53	0.52
E_{NS}	Unit less	1	0.68	0.69	0.67	0.66	0.55	0.60	0.57	0.54
DRMS	mg l^{-1}	Smaller value indicates a better model performance	1.95	0.98	2.03	0.83	1.76	1.15	2.07	0.94
PBIAS	%	0 (+ values = bias to underestimate & - values = bias to overestimate)	2.33	-3.73	2.65	-9.07	4.22	-6.58	4.13	-2.56

6.6 Model Calibration and Validation Discussion

An overall weakness of the SWAT model is the use of equations that have parameters that are not directly measured. For example, the curve number equation, although used often to estimate run-off volumes, is highly uncertain due to the use of a parameter (i.e. the curve number) that had not been determined empirically for the UK but rather for the USA using different land uses. In addition, the MUSLE, which is used for soil erosion simulation, is also uncertain because of the number of parameters in the equation that are set from qualitative information (e.g. soil type and ground cover). Efforts have been made to incorporate more process-based equations; there is still room for improvement in some of the basic processes modelled by SWAT. Some limitations of the SWAT model have been noted in the following section where the main aim is to discuss the limitations of the input and calibration data for the SWAT model.

6.6.1 Soils Investigation

In Chapter Five the use of the National Soil map in the SWAT model was discussed. The map includes the soil associations and does not incorporate the soil series, which make up the association. Soils that are highly erodible could be missed from the model giving unreliable results. To investigate this, USLE calculations were undertaken for each soil series within each sub-basin. Results showed that soils such as Sheringham, which make up approximately 28% of the Wick 3 series have a high mean annual soil loss, but due to the current modelling of soil associations with SWAT are not represented in the model.

To investigate soil erodibility further a sensitivity analysis has been carried out for the distribution of the soil associations and their sub-groups on a small sub-basin within SWAT. The sub-basin was set up within SWAT using calibrated parameters from the Bure and Ant SWAT model. It has been modelled with three different soil scenarios (Table 6.6.1); each soil scenario was simulated in SWAT with 3 different land cover types (winter wheat, maize and pasture). The Sheringham soil series was modelled; along with the three different land cover types to verify whether SWAT is responding in an appropriate way to different soil and land covers (the Sheringham association is not found within the study area, it is only an ancillary soil to the Wick association).

Table 6.6.1: Soil investigation scenarios

Scenario	Soil
One	Wick series covering whole sub-basin
Two	Wick ancillary. Sub-basin split into 4 HRU’s: Wick (38% of subasin), Wickmere (36%), Sheringham (16%), Aylsham (10%)
Three	Sheringham series covering whole sub-basin

Figure 6.6.1 shows that as expected when maize is used as the land cover type it will result in a higher sediment yield, as maize is recognised as having a higher risk of soil erosion and run-off than most other crops (DEFRA, 2005). Sediment yield is highest

for Sheringham soil for all three-cover types. This is also as expected as the Sheringham soil has the highest USLE mean annual soil loss (3.46 t ha^{-1}). SWAT is responding in a realistic way to different soil and land cover combinations.

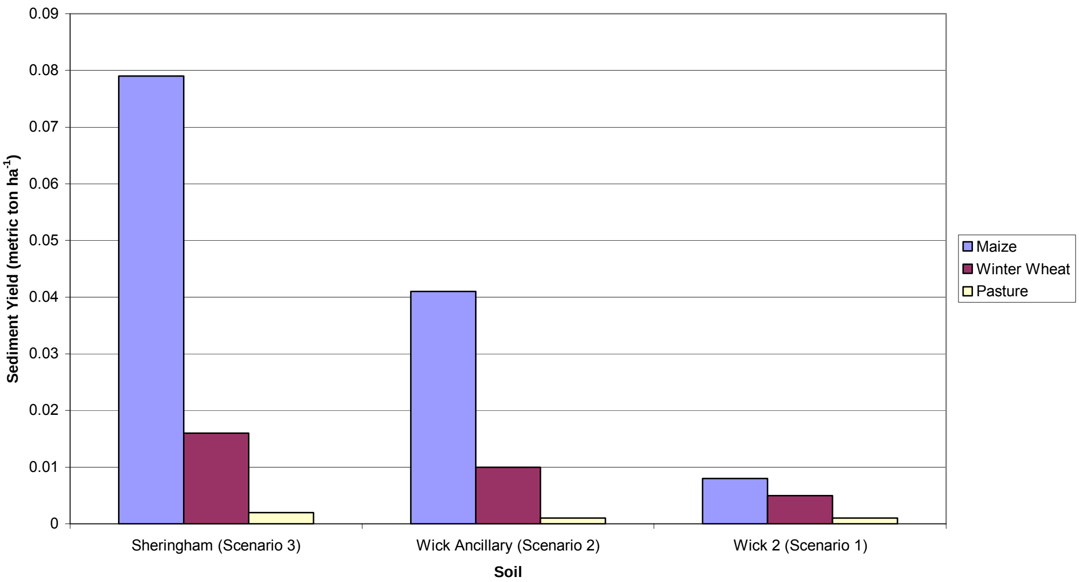


Figure 6.6.1: Average annual sediment yields for varying soil and land cover types

The difference between sediment yields for the three soil scenarios when modelled with maize is quite significant. The difference between the three soils when used as pasture is relatively small.

Figure 6.6.2 shows that the difference in sediment yields for Wick 2 and the Wick ancillary soil scenarios when modelled with winter wheat is very small. Only in 1991 is the sediment yield significantly larger for the Wick ancillary scenario. A difference between the two Wick soil scenarios was expected as the Wick ancillary scenario incorporates the Sheringham soil series. The difference between the two soils in terms of sediment yield is relatively low and the use of the National soil map and Soil associations has a limited influence on SWAT sediment yield results. Research carried out by Di Luzio *et al.* (2005) also showed that although land use and land cover maps had a significant effect on sediment yield prediction; soil maps had an insignificant influence.

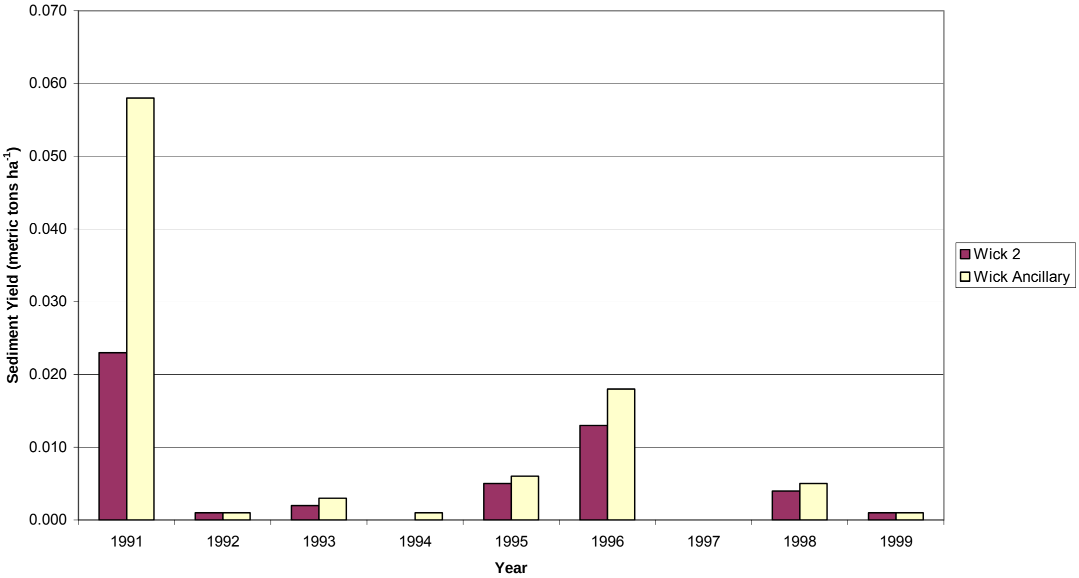


Figure 6.6.2: Sediment yield for winter wheat

6.6.2 Hydrology

Calibration and validation results show that SWAT is reasonably able to predict total and base flow within the Bure and Ant watersheds, with all percentage errors falling below published criteria. The Nash and Sutcliffe efficiencies for all three calibration sites are also comparable with other UK SWAT models. Kannan (2004) achieved E_{NS} values of 61.20% and 59.57% whilst White *et al.* (2004) achieved an E_{NS} of 53% for the Wensum watershed (adjacent to the River Bure watershed).

In comparison to studies carried out in other countries the E_{NS} values attained in the UK are relatively low. Jha *et al.* (2003b) obtained an E_{NS} value of 93% in Raccoon watershed in Iowa, USA. In Spain Conan *et al.* (2003) obtained a value of 72% whilst in Sardinia a value of 83% was achieved (Cau, *et al.,* 2003). The most obvious difference in these SWAT models to those built in the UK is the size of the watersheds being modelled. The Bure and Ant model covers approximately 612 km^2, the Roccoon model in Iowa covered 9,500 km^2, the Guadiamar watershed in Spain covers 1,500 km^2 and the Sardinia model is 24,089 km^2. These results illustrate a possible limitation in the SWAT model when modelling small watersheds. However, within the above mentioned studies no real attention has been made to the internal working of the model.

Due to the small size of the Bure and Ant catchment the response of the watershed to precipitation events is more sensitive and the timing of events is crucial for model performance. Consequently, errors that may be ‘averaged out’ on larger basins would be quite apparent on small basins as the input data and parameters have been set on a large scale. This is due to the fact that parameters had to be adjusted on a basin wide basis due to small number of gauged sites in the two watersheds. This is especially so in the Ant watershed where there is only one gauged site (Honing Lock) near the headwaters of the watershed.

The highest percentage errors were attained in the spring and summer months, which represent the lowest flows in the year. The daily time series graph for Ingworth shows that SWAT is over predicting the low spring and summer flows (Fig 6.6.3). This error accumulates in SWAT, increasing in sub basins further down watershed (Fig 6.6.4).

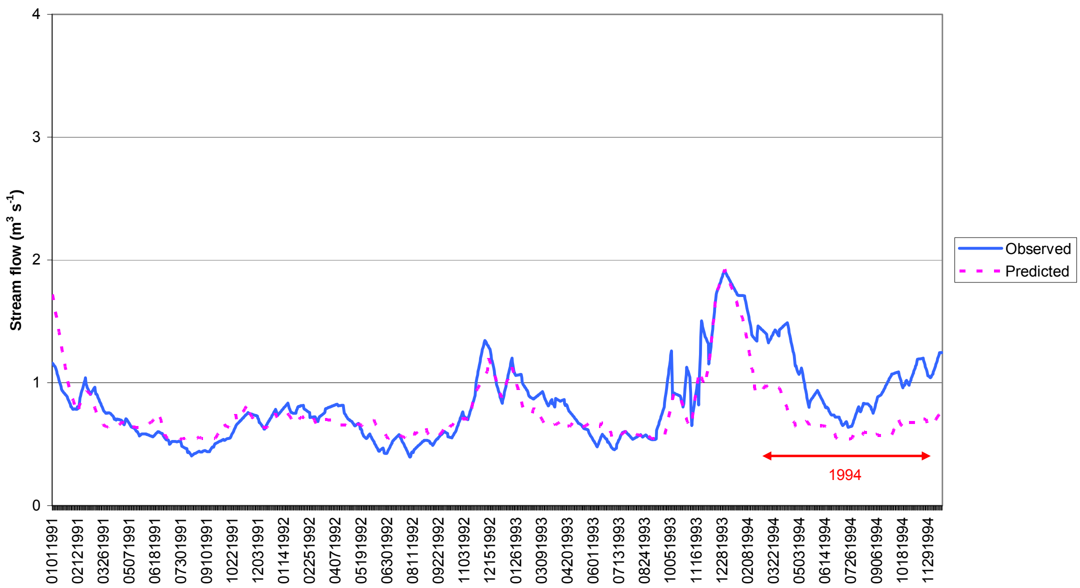


Figure 6.6.3: Base flow at Ingworth (1991 – 1994)

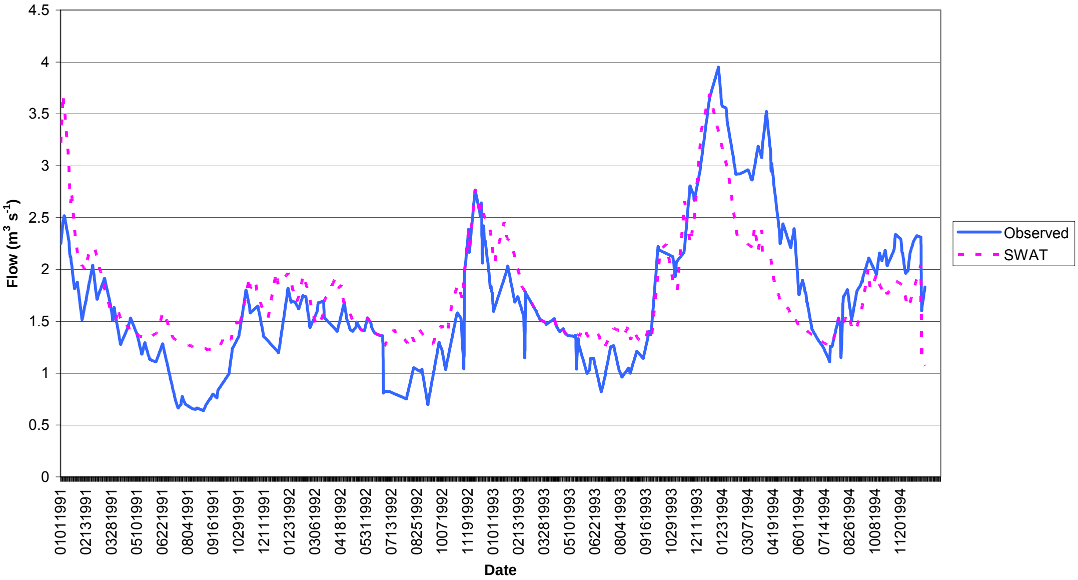


Figure 6.6.4: Base flow at Horstead Mill (1991 – 1994)

It is possible to reduce the spring and summer flows predicted by SWAT by increasing the SWAT GWQMN value within the ground water files. By increasing this value a higher portion of rainfall appearing as base flow will be retarded and stored in the soil. Although this will reduce the spring and summer base flow it will reduce base flow through out the rest of the year. It will also increase the amount of water being stored in the soil.

The amount of water being stored in the soil has been shown to be realistic as described in section 6.2. Figure 6.6.4 also shows that in the winter months SWAT is generally under predicting base flow, decreasing base flow further to improve the spring and summer flows will increase this under prediction. Increasing the soil water being stored and decreasing base flow through increasing the GWQMN is not an option.

To solve this problem the input data need to be looked at. Spring and summer low flows within the Bure and Ant watersheds are predominately made up of point source discharges such as flows from STW's. These have been incorporated into the model through Environment Agency licensed discharge values. When compared to Anglian Water discharge values for three STW in the Bure and Ant watersheds it was found that the EA licensed values were lower than actual recorded discharge values for the three sites observed by Anglian Water (Fig 6.6.5). Maximum recorded values by Anglian Water are also substantially higher then those consented by the EA, particularly at Belaugh STW.

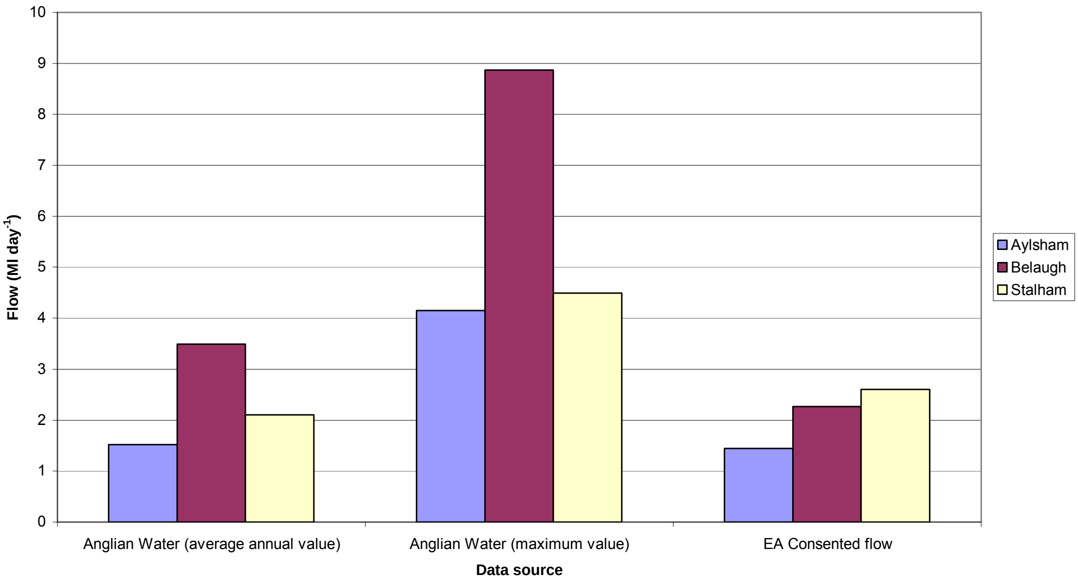


Figure 6.6.5: Comparison between actual STW discharge flows and EA consented flows

Anglian Water data show that discharge values vary throughout the year (Fig 6.6.6). Lower discharge flows can be seen in the summer months, when SWAT is over predicting summer flows. Higher flows are occurring in the winter months, where SWAT is under predicting base flow. EA consented flow data are given as just one value for the whole year. This has been input to SWAT as an annual point source discharge value. SWAT is not predicting the variability, which is shown in the actual flow data. It is possible to input monthly and daily point source discharge values into SWAT however no long-term data from Anglian Water are available, therefore annual EA consented values have had to be used.

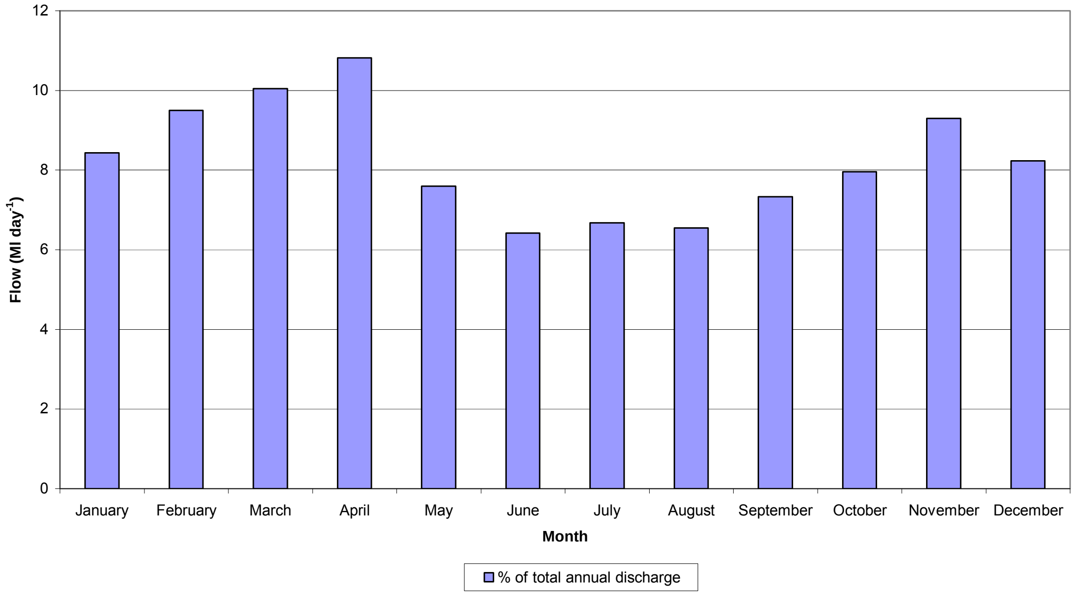


Figure 6.6.6: Daily flow at Aylsham STW

Although STW discharge may go some way to explaining the discrepancy between separated base flows and predicted values, the variation between monthly STW discharges is only 2 Ml day^{-1}. In comparison the total discrepancy between separated and predicted base flow is 5 Ml day^{-1}. The remaining discrepancy may be due to the uncertainties in crop and soil distribution and irrigation application as these all affect contributions to base flow.

The Environment Agency has also provided the abstraction data used in SWAT. This has been input to SWAT through the consumptive water use (WUS) file. This removes water at the sub-basin level from the reach, shallow aquifer or deep aquifer. Within each sub-basin there are a number of abstraction points recorded by the EA; these have had to be lumped together as SWAT only allows one abstraction point to be modelled. It will not take into account change in abstraction values over time.

Figure 6.6.3 shows the base flow at Ingworth. It can be seen in winter 1993 – 1994 that there is a significant decrease in SWAT base flow values compared to separated observed base flow values. This carries on into the validation period (Fig 6.6.7) and is especially apparent in the winters of 1995 – 1996 and 1998 – 1999.

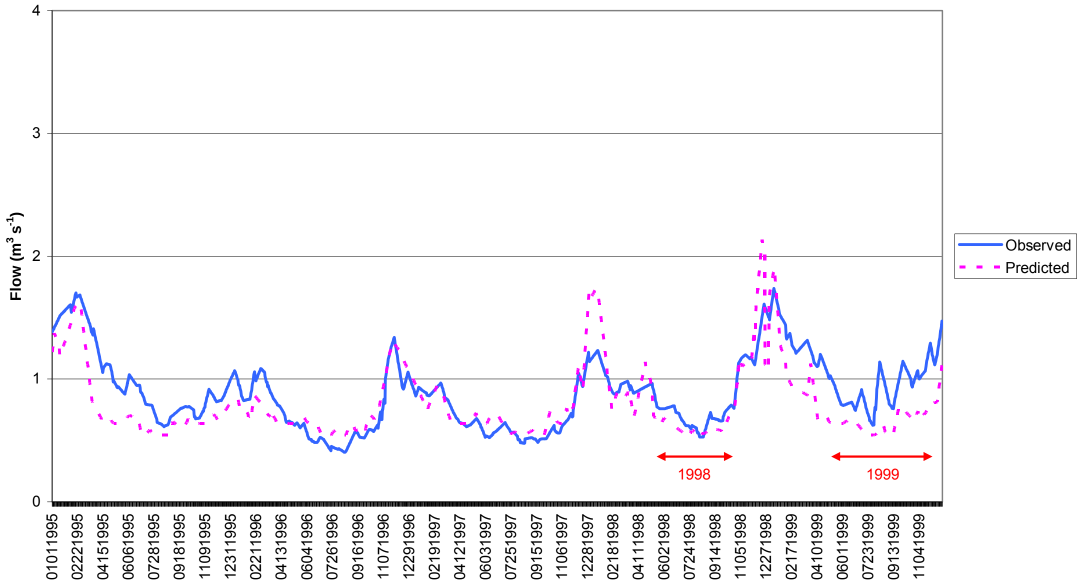


Figure 6.6.7: Base flow validation values at Ingworth (1995 – 1999)

Table 6.6.2 shows the number of abstraction licences revoked in 1993. A total of 21920 m^3 day^{-1} was not removed from the system from 1993 onwards. This is approximately 5% of the total volume of water, which is abstracted from the model. It is not possible to vary abstraction over time in SWAT, therefore it is not possible to achieve accurate base flow values and this is represented in the E_{NS} value and higher percentage errors for seasonal variability. At the end of both the calibration and validation period (years 1994 and 1999), base flow decreases dramatically throughout the whole year, which can not be sufficiently explained by changes in abstraction.

It is suggested that this is a function of percolation to the deep aquifer. Movement to the deep aquifer occurs only if the movement of water stored in the shallow aquifer exceeds the threshold value specified by the user (REVAPMN). The calibrated REVAPMN threshold was set to 1mm; recommended values can lie between 0 and 500mm. The value is sufficient for the majority of the calibration and validation period; the years 1994 and 1999 are both wet years with total rainfall falling above the annual average of 520.6 mm yr^{-1} (587 mm yr^{-1} and 616.7 mm yr^{-1} respectively). By having a low REVAPMN threshold value during wet periods excess water is percolated into the deep aquifer where it is stored instead of contributing to base flow. By increasing the REVAPMN value this will adversely affect the base flow calibration for years with average and below average rainfall, where base flow is already being underestimated in the winter months.

Table 6.6.2: EA abstraction licence revoked in 1993

Sub-basin	Revoke date	Amount (m^3 day^{-1})	Site name
1	01/04/1993	1320.00	BORE AT BESSINGHAM
1	01/08/1993	1800.00	WELLPOINTS AT EAST BECKHAM
1	01/08/1993	1200.00	TRIB W/C OF SCARROW BECK
3	01/07/1993	8.00	BORE,WOOD FARM,EDGEFIELD
4	01/05/1993	2.30	BORE AT DAIRY FARM,HEYDON
4	01/06/1993	13.20	WELL,MOOR HALL FARM,BRISTON
4	01/06/1993	1.10	WELL AT HEYDON
4	01/04/1993	215.00	R BURE AT SAXTHORPE
4	01/07/1993	660.00	R BURE,BINTRY FM,ITTERINGHAM
4	01/07/1993	215.00	RIVER BURE AT CORPUSTY
6	01/06/1993	13.00	BORE,SEAMAN'S FARM,GUESTWICK
8	01/09/1993	1450.00	BORE,CHURCH FARM,OULTON
9	01/04/1993	19.90	RESERVOIR AT COLBY
9	03/04/1993	900.00	BORE,HELSDON FARM,HANWORTH
9	04/04/1993	1455.18	BORE,HELSDON FARM,HANWORTH
9	04/04/1993	1455.18	RESERVOIR NO 1,ROUGHTON
9	04/04/1993	1455.18	RESERVOIR NO 2,ROUGHTON
12	01/06/1993	2012.00	BORE,OAKS FARM,FELMINGHAM
12	01/04/1993	636.00	RES AT SUFFIELD
12	01/04/1993	636.00	SUFFIELD BECK AT SUFFIELD
14	01/04/1993	1500.00	BORE AT GRANGE BUILDINGS D'HAM
14	01/04/1993	1905.00	BORE AT MANOR FARM DILHAM
17	01/04/1993	32.00	AGRIC BORE,HALL FARM,OXNEAD
17	01/04/1993	636.00	IRRIG BORE,HALL FARM,OXNEAD
17	01/04/1993	636.00	WELL AT HALL FARM,OXNEAD
19	01/11/1993	1744.00	GRAVEL PIT – HEVINGHAM
	Total	**21920.04**	

Although recommended calibration methods were followed some of the calibration parameters were found to make minimal change in model output. Recommended variables for calibration of temporal patterns of stream flow are hydraulic conductivity and base flow alpha factor. Changing the hydraulic conductivity caused monthly averages to increase or decrease by comparable amounts for all months. The most effective parameters were those to do with groundwater flow (groundwater re-evaporation coefficient, minimum depth of water in soil for base flow to occur and minimum depth of water in shallow aquifer for re-evaporation).

6.6.3 Sediment

Sediment erosion from each HRU is simulated using the Modified Universal Soil Loss Equation (MUSLE) (Williams and Berndt, 1977). This equation replaces the traditional Universal Soil Loss Equation (USLE) rainfall factor with a run-off factor. The MUSLE is solved for each HRU and final sediment yields are routed down the main channel using a stream power equation (Neitsch *et al.,* 2001). This routing method assumes the maximum amount of sediment that can be transported in a given reach is a function of the peak channel velocity (Arnold *et al.,* 1995b).

Sediment erosion and transport modelling is highly uncertain and accurate simulation of sediment processes on the land surface, is difficult to capture due to the heterogeneous nature of a watershed and the relatively unrefined equations used to explain certain processes (e.g. MUSLE). As a result, it is typically the case that a model that performs acceptably well for hydrology may still have limitations in fully capturing sediment loads. This is the case with the Bure and Ant SWAT model.

Due to lack of data, calibration was concentrated at two sites on the River Bure (Scarrow Beck and Wroxham) and validation was carried out on one site on the River Ant (Honing Lock). Scarrow Beck is close to the headwaters of the River Bure and is a small sub-basin. SWAT is predicting an almost constant sediment concentration, picking up none of the variability that is shown in the observed data. Further down stream at Wroxham Rail Bridge where the sub-basin drains a larger area the suspended sediment concentrations predicted by SWAT match the observed data slightly better. SWAT especially picks up well the lower sediment values in the summer months, but over predicts sediment in the winter months. However, SWAT does not pick up any of the variability in sediment values for the validation site at Honing Lock, as a result of poor surface flow representation.

The variation in results between calibration/validation sites may be due to the sub-basin size and location within the modelled watershed. Binger *et al.* (1997) found that sediment yield varied significantly with changes in sub-basin size and location. These effects were attributed to increasing levels of aggregation on average sub-basin slope and on the proportion of the sub-basin delineated as cropland. FitzHugh and Mackay

(2000) also demonstrate that sediment generation is attributed not only to land cover and soil but also to the area of a sub-basin. They show a non-linear relationship between sediment generation and the size of the sub-basin. If this is the case then run-off should respond in a similar way as hydrological data is also aggregated at the HRU and then sub-basin level. In his work Binger *et al.* (1997) also showed that run-off is not sensitive to sub-basin or HRU size. It can be said that the aggregation effect observed is partly due to model structure and hence the sub-models for run-off and sediment yield respond differently to the same change in spatial representation.

Monthly and bi-monthly observed data were used to calibrate the SWAT model. The observed data used in validation is daily data from Johnes (1996b), which accounts for the increased variability in observed data compared to the calibration sites. By using daily data to validate the sediment loads in SWAT it can be seen that SWAT cannot capture sediment erosion during individual storm events. The MUSLE algorithm is designed to simulate erosion occurring due to the run-off produced during storm events (Williams and Berndt, 1977). Arnold *et al.* (1998) stated that ‘SWAT does not simulate detailed event based flood and sediment routing’. They felt the model was best developed to evaluate management impacts on long-term erosion and sedimentation (Arnold *et al.,* 1998). It would be inaccurate to apply the model in an attempt to evaluate particular storm based events. It also demonstrates that daily observed sediment loads can not reliably be used to calibrate and validate a SWAT model.

A greater seasonal variability may be picked up with SWAT suspended sediment values if all sources of sediment within the Bure and Ant watersheds were captured within the model. Sediment within the Bure and Ant watershed can be generated by three main sources; land erosion, bank erosion from tides and boating activities and finally from STW’s. Although not a source of sediment the re-suspension of sediment due to boats and tidal influence in the rivers greatly impacts suspended sediment concentrations within the Bure and Ant watersheds.

SWAT is able to model surface run-off through the use of the SCS curve number method. Sediment generation from each HRU is then calculated using MUSLE. Although it is possible to model bank erosion within SWAT through the channel erodibility factor no measurements are available to estimate the parameter accurately, so it has been set to 0.5. A value of 0.0 indicates a non-erosive channel while a value of 1.0 indicates no resistance to erosion. Although the erodibility factor is considered within SWAT the sensitivity of the parameter is questionable as the sediment routing routine, which has relatively simplistic equations (Arnold *et al.,* 1998), does not consider important sediment transport characteristics such as bottom shear stress. This determines whether erosion or deposition will occur, given flow velocities and resulting shears.

It is possible to model sediment loads from STW's. As for effluent, sediment discharge from STWs can be modelled as a point source at either a daily, monthly or annual level. It is very difficult to obtain comprehensive sediment data on locations, consented and/or actual discharges from sewage treatment works in the Broads catchments (White *et al.,* 2005). These data have not been incorporated into the model.

Within the Bure and Ant model the only source of sediment is from land. Land use significantly affects the magnitude of sediment loss through its influence on the degree of the protection afforded by the vegetation cover. The relationship between sediment yield and annual surface run-off can be seen in Figure 6.6.8 for the major soil association in the study area (Wick). It can be seen that agriculture is the land cover type that produces the highest sediment rates and that SWAT is predicting run-off sources of sediment well. Sediment representation within SWAT could be improved if more data were available.

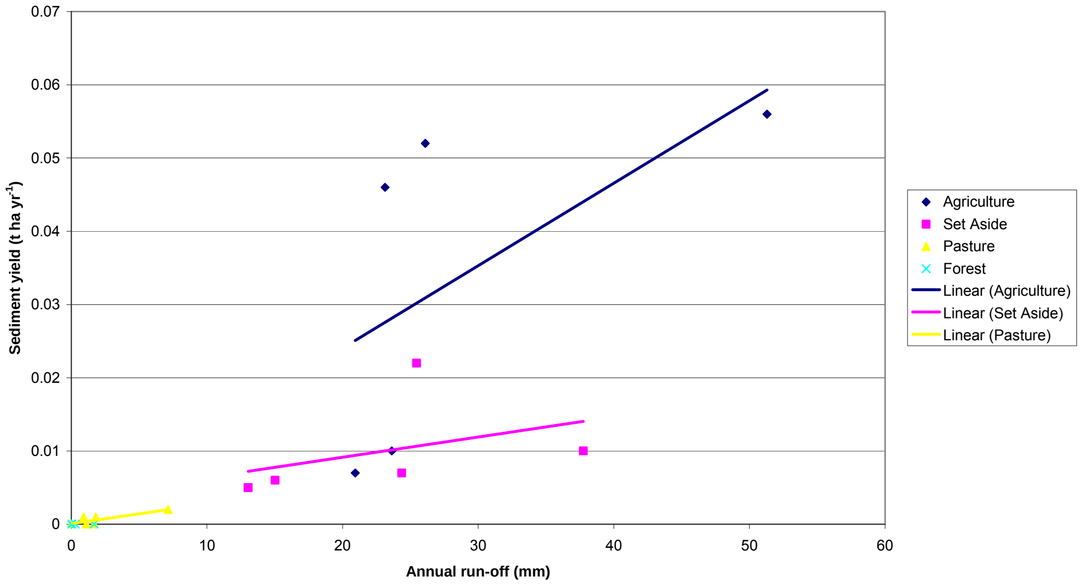


Figure 6.6.8: Relationship between modelled sediment yield and annual run-off (1991 – 1995)

6.6.4 Nutrients

Overall calibration for nutrients was reasonable, calibration of nitrate within the model was good at all sites, problems did occur with total phosphorus. The modelled phosphorus concentrations demonstrated much more variability than seen in the monitored data. Phosphorus moves in a much more dynamic event based way than nitrate and such variability is to be expected at a daily level.

Total phosphorus calibration was undertaken using Environment Agency monthly measured data; validation was partly done using weekly and daily data. It can be seen from the graph below that SWAT is able to predict total phosphorus reasonably well when looking at long term trends in comparison to monthly observed data. The models performance is ideal for evaluating management impacts on long-term erosion and water quality within the Bure and Ant watersheds.

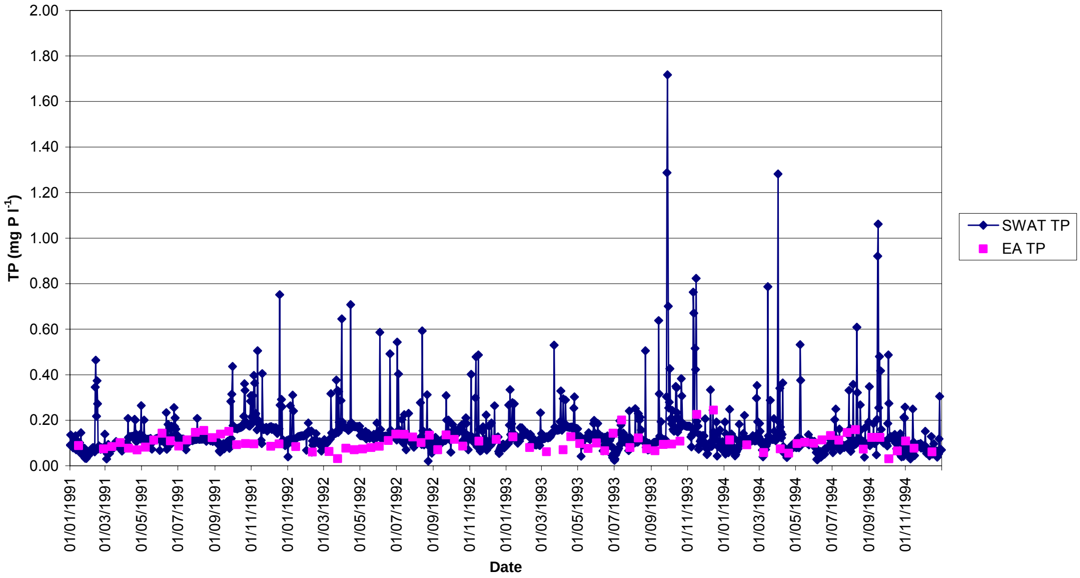


Figure 6.6.9: Long term phosphorus trends at Wroxham Rail Bridge

When considering phosphorus dynamics on a daily basis SWAT is not able to predict total phosphorus concentrations with great accuracy (Fig 6.6.10).

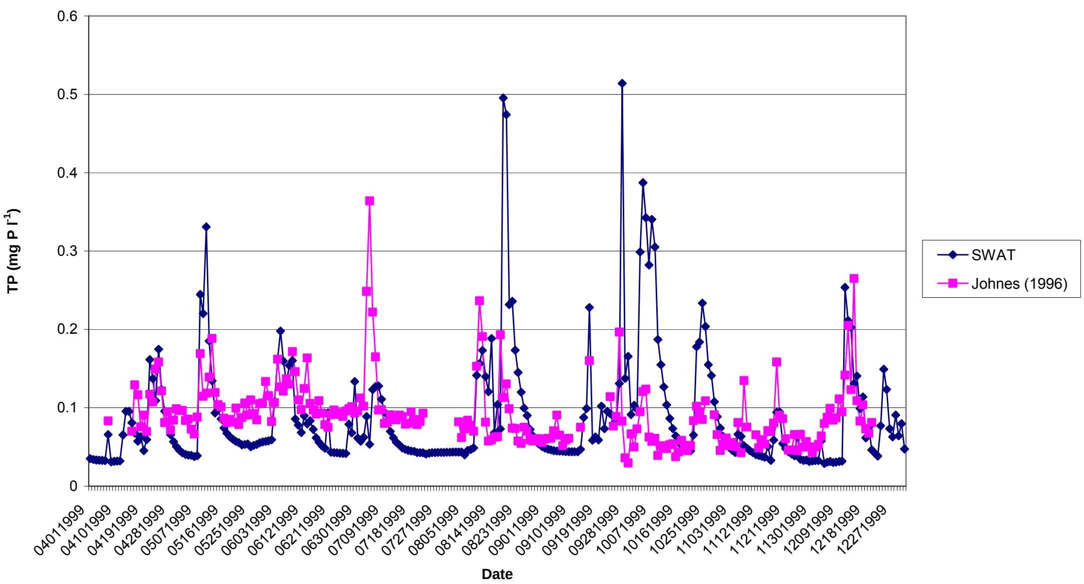
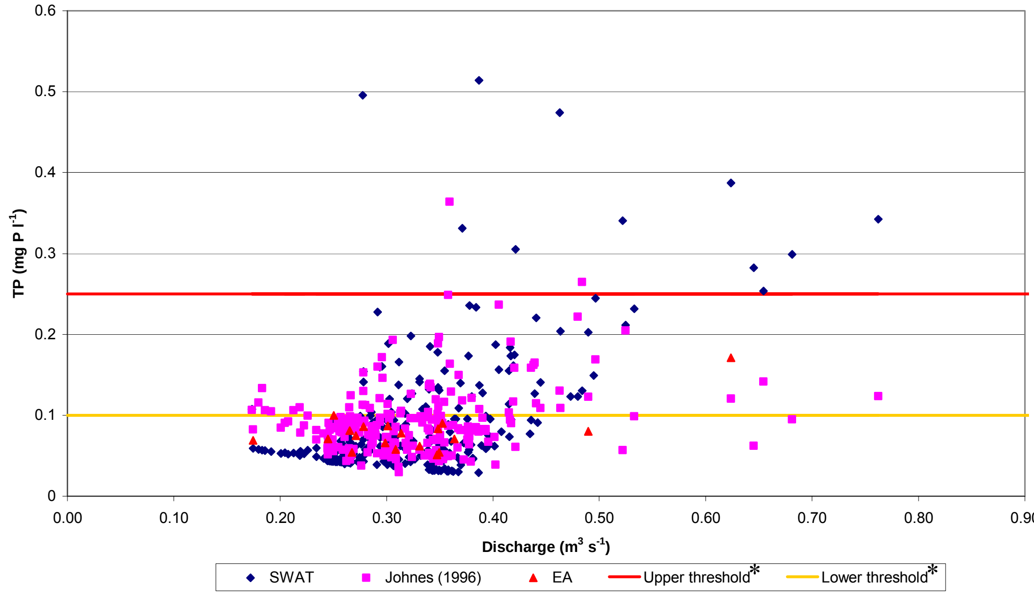


Figure 6.6.10: Comparison between SWAT and daily observed total phosphorus concentrations at Honing Lock

SWAT overestimates peaks and the timing of peaks when compared to daily observed total phosphorus concentrations. SWAT is also predicting many days with the same total phosphorus values, giving an almost smooth line for long time periods. Johnes (1996b) daily data clearly shows that total phosphorus concentrations are very dynamic, changing on a daily basis. SWAT is unable to predict such variability for two main reasons. It was discussed previously that SWAT is unable to predict event based storm, sediment or water quality values. Limited data were available for actual STW phosphorus discharge rates. Only one consented value was provided by the EA and 2 published average values were available. Only constant STW phosphorus could be input into SWAT as point source discharges resulting in the almost smooth line seen in Figure 6.6.10. Therefore although daily data area available for calibration SWAT performs better when comparing predicted total phosphorus values to either monthly or weekly observed data.

The Environment Agency only measure water quality parameters on a monthly basis. With this time resolution high discharge events are under estimated and therefore the flux of sediment and nutrients are also underestimated (Fig 6.6.11). This is especially so in the Broads, as the flat landscape means that the rivers have a very low channel gradient. In combination with low rates of discharge this means that the rivers are slow flowing, with high sediment trapping efficiency. Much of the nutrient load exported from diffuse sources in Norfolk; in particular particulate phosphorus is trapped in the bed sediments of the rivers during base flow conditions. This trapped sediment is not static and during storm flow, is re-suspended and transported downstream to the tidal reaches where diurnal tides keep much of the particulate nutrient load in suspension. This means that in the fresh water river basins in the Norfolk region, current water quality monitoring schemes operating on a monthly basis will be systematically underestimating nutrient loads, especially phosphorus which is largely exported in particulate form from agricultural sources.



* Ecological failure criteria have been defined by Severa-Martinez (2005) in terms of total phosphorus an ecological threshold of 0.25 mg l^{-1} (upper) - 0.1 mg l^{-1} (lower). TP concentrations above these values will have an adverse effect on ecology.

Figure 6.6.11: Comparison between daily and monthly recorded phosphorus concentrations at Honing Lock

The implications of under estimating nutrient loads can be seen in Figure 6.6.11. Severa-Martinez (2005) defined an ecological failure criterion for Hickling Broad. In terms of total phosphorus a maximum ecological threshold limit of 0.1 – 0.25 mg l^{-1} was suggested; total phosphorus concentrations above these values will have an adverse effect on Hickling Broad. The lower TP threshold (0.1 mg l^{-1}) also represents the EA river water quality target for the study area under the General Quality Assessment Scheme (EA, 2004). The upper and lower threshold limit have been plotted in Figure 6.6.11, from the graph it can be seen that if only EA monthly data are considered that the lower threshold limited is only breached once in 1999, therefore no adverse affects on Hickling Broad would be expected. If Johnes (1996b) weekly and daily data are considered, then the lower threshold limit is breached 61 times i.e. 17% of time and the upper threshold is breached twice in 1999. If SWAT data are then looked at both thresholds are breached 20% of the time, having an adverse effect on Hickling Broad, which would be underestimated or missed if only monthly data were considered. Work carried out by Leecaster and Weisberg (2001) also observed percentage changes in events in which standards were exceeded depending on sampling frequency. It was found that sampling at three times per week resulted in the observation of 55% of the events in which standards were exceeded. This frequency dropped to 25% and 5% for weekly and monthly sampling respectively.

Johnes (1996b) and SWAT data show that it is at higher flows that higher phosphorus concentrations are likely. EA data are predominantly collected at lower flows as can be seen in Figure 6.6.11. Work carried out by USGS (2001) suggests that the best overall monitoring strategy for accurate and precise load and trend estimations of sediment and nutrients consists of 50% base flow samples and 50% storm samples. At Honing Lock the average base flow is 0.25 m^3 s^{-1} and the average peak flow is 0.35 m^3 s^{-1}. From Figure 6.6.12 it can be seen that, for the year considered only 3 EA samples are taken at peak flows, this is only 25% of the time for monthly samples. Based on monthly samples the ability to ensure adequate sampling of all river stages is severely limited. Zhang (1998) stated that for river basins less than 1000km^2 (which all the river basins in the study area are) storm event sampling should be carried out wherever possible as this reduces the random possibility of sampling high flow by constant frequency sampling. By increasing sampling frequency to either bi-monthly or weekly the percentage of samples taken at high flows would increase to cover the majority of peak flows but would perhaps still miss extreme peak flows as can be seen by Johnes (1996b) data ranges.

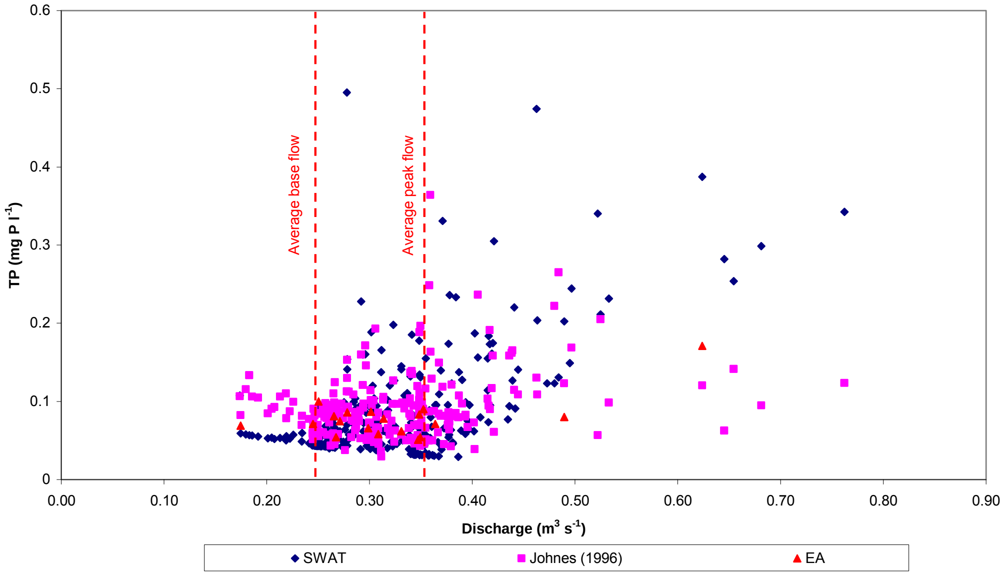


Figure 6.6.12: Comparison between daily and monthly recorded phosphorus concentrations for different flow rates at Honing Lock

Figure 6.6.13 and the above discussion show that the fewer number of samples the greater the error in sample results and that with an increased sample size a more representative data set may be achieved. Currently the EA are only taking monthly samples, a total of 12 samples a year are taken. A more suitable sampling frequency can be calculated using a simple statistical formula.

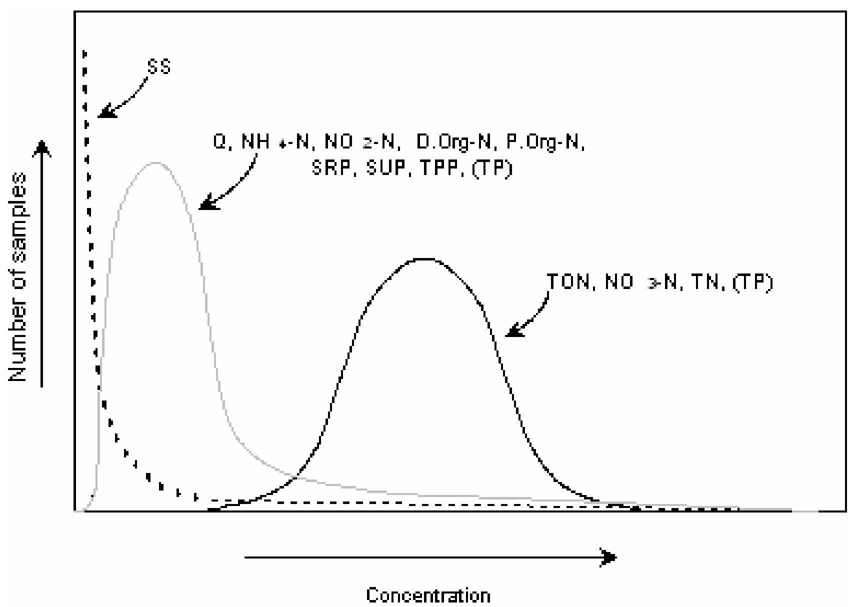


Figure 6.6.13: Hydro chemical response characteristics of flow, SS and nutrients in UK Rivers

Sample size can be calculated using the following formula:

$S = (z/e)^2$

Where:	s	the sample size
	z	degree of confidence (95% confidence is most frequently used and accepted). When using a 'z' table the value of 'z' should be 1.96 for 95% confidence.
	e	accepted error measured as a proportion of the standard deviation (accuracy).

A sample size to aim for in order to be 95% confident in the result, with an error of 30% of the population standard deviation (SWAT results show a standard deviation of 0.07 in total phosphorus results for all sites, an accepted error of 0.021 (30%)) results in the following calculation:

$s = (1.96 / 0.3)^2$

Therefore s = 42.68

In other words, 43 samples a year would need to be taken to meet the criteria. This results in a weekly sampling programme. This value will increase if a lower standard error is required.

It can be seen from Figure 6.6.11 and 6.6.12 SWAT is able to predict total phosphorus concentrations at varying flows well when compared to the daily and weekly data of Johnes (1996b), even though SWAT was calibrated on monthly EA nutrient data. It is not able to predict daily concentrations with great accuracy as discussed previously in this chapter when calibrated on monthly data. Therefore for SWAT modelling weekly samples are sufficient for calibration and validation purposes. If more detailed knowledge of nutrient dynamics is required in the future then it would be advisable to undertake weekly, daily, sub-daily samples or event-based samples as discussed above.

On the basis of the nutrient calibration results it was concluded that SWAT has a shortcoming in nutrient calibration due to the sensitivity of the model to many parameters. Changing the sensitive parameters for phosphorus was affecting the calibrated nitrate levels. This is because the same parameters (crop residue coefficient and bio-mixing efficiency) can be used to calibrate both phosphorus and nitrate within the model.

The calibration and validation procedure undertaken indicates limitations in the predictive capability of the model, especially for sediment. There are many possible sources of these errors, which have been discussed: lack of input data, over simplification of various factors in the model equations, non-optimal calibration parameters and errors in observed output data. Based on current available data, the model demonstrates its utility as a tool to understand processes in the watershed and as a basis for effective management in the Bure, Ant and Upper Thurne watersheds.

Chapter Seven Upper Thurne SWAT Model

7.0 Upper Thurne Model Build

The same methodology used in the Bure and Ant SWAT model build was followed to build the Upper Thurne model. Crop rotations established from EDL data and ADAS standard rotations within the Bure and Ant model have again been used to ensure a good representation of EDL data within the SWAT model (Fig 7.0.1). The use of the Bure and Ant crop rotations has meant the easy transfer of management files between the two SWAT models.

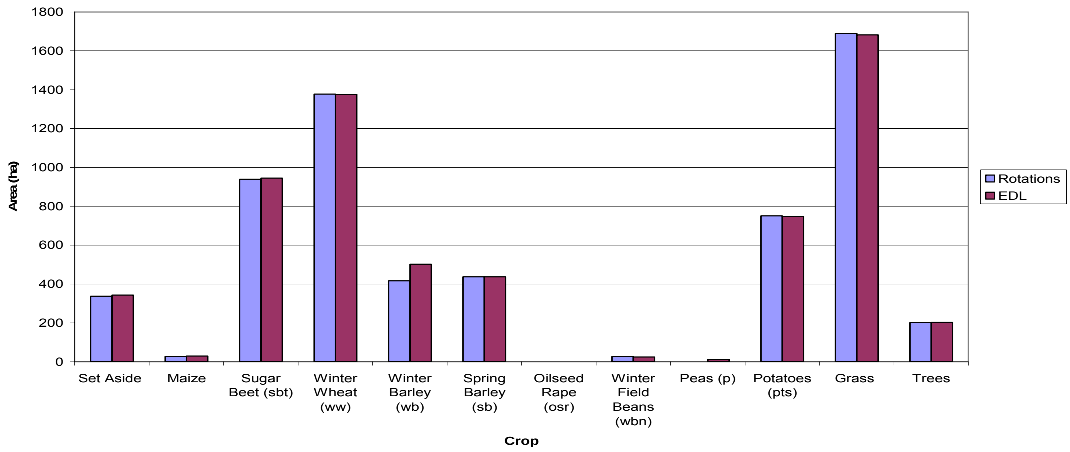


Figure 7.0.1: Comparison of modelled and actual crop areas within the Upper Thurne

As no flow or sediment data are available for the calibration of the Thurne model calibrated base flow parameters from the Bure and Ant model have also been transferred to the Thurne model along with calibrated sediment parameters (Table 7.0.1).

Table 7.0.1: SWAT Bure and Ant calibrated parameters used in the Upper Thurne model build

Parameter	SWAT Value Ranges	Calibrated Value
ALPHA_BF	0.1 – 0.3 for a slow response watershed 0.9 – 1.0 for a fast response watershed	0.10
GWQMN	0 – 5000 Set by user	10
GW_REVAP	0.02 – 0.2	0.2
REVAPMN	0 – 500 Set by user	1.0
RCHRG_DP	0.0 – 1.0	0.7
GWHT	0 – 25 Set by user	1.0
BIOMIX	0.00 – 1.00 Default 0.2	0.01
FRT_LY1	Default 0.00	0.001
SLSUBRSN	Varies for each HRU	20% reduction in value set by SWAT
SLOPE	Varies for each HRU	10m reduction in slope length value set by SWAT
CN	Varies for each land cover	10% increase in value set by SWAT

7.1 Land Drainage Pump Modelling

The role of the pump and dyke system in the Thurne river basin is to keep the broad land drained so it may be used as permanent agricultural land. Originally sluice gates were placed on the dykes so that when the flap sluice at its outfall was tide locked, water could be temporarily stored. This would reduce the risk of the dyke banks being breached following periods of heavy run-off, with consequent flooding of the adjoining marshes (George, 1992). Sluice gates have now been replaced by electric pumps but still serve the same purpose to temporarily store water.

It is not possible to model pumps within SWAT as there is no function for this. Therefore a conceptual model of the Upper Thurne broads and the Internal Drainage Board drainage system (Fig 7.1.1) has been produced to assess the best way of modelling the pumps within SWAT. From the diagram and Figure 7.1.2 it can be seen that each pump has an IDB drain upstream of it, while the pump is not in operation water is stored in the upstream dykes. Once the pump is in operation water is pumped into a broad or the River Thurne. Two possible solutions can be seen; either to model the pumps as a point source discharge into the system at the top of the river basin, or to model the pumps and their up stream dykes as reservoirs. By modelling the pumps as point source discharges this will not represent the temporary storing of water in the dykes or the change in flow into the broads, which occurs with the turning on and off of the pumps, as only a constant flow can be added. Both of these factors affect water quality, especially the sediment load reaching the Broads within the Upper Thurne system. The transfer of water from the Stubb Mill pump to Brograve cannot be sufficiently modelled with either solution; therefore it has been treated as a separate input to the system.

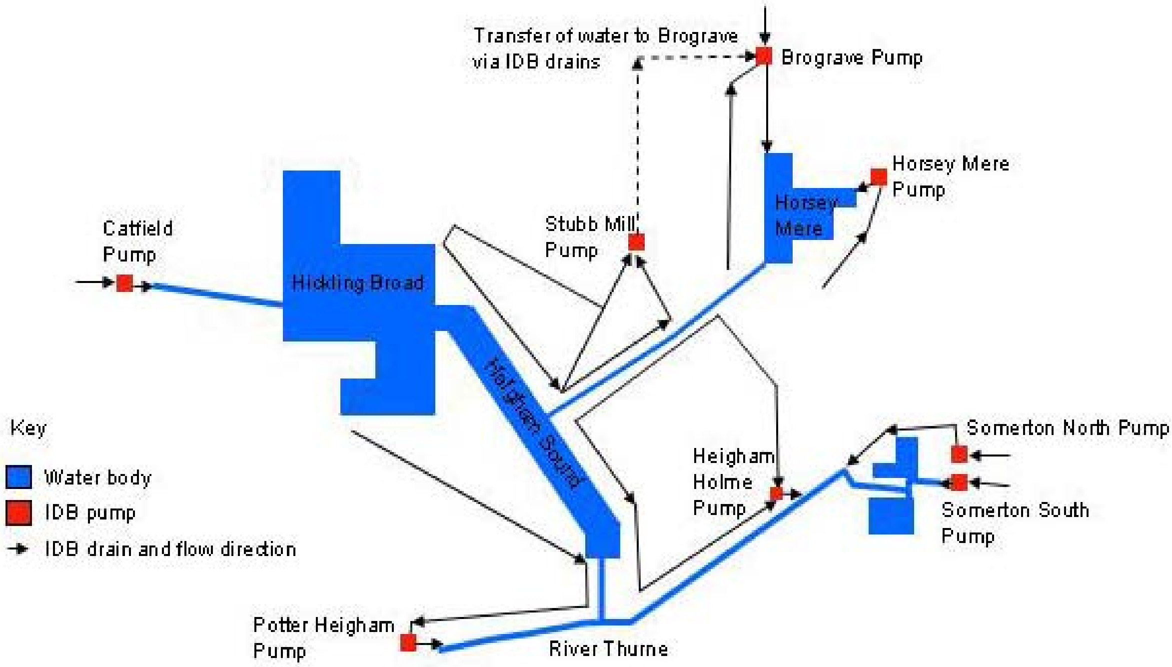


Figure 7.1.1: Conceptual model of the Upper Thurne IDB drainage system

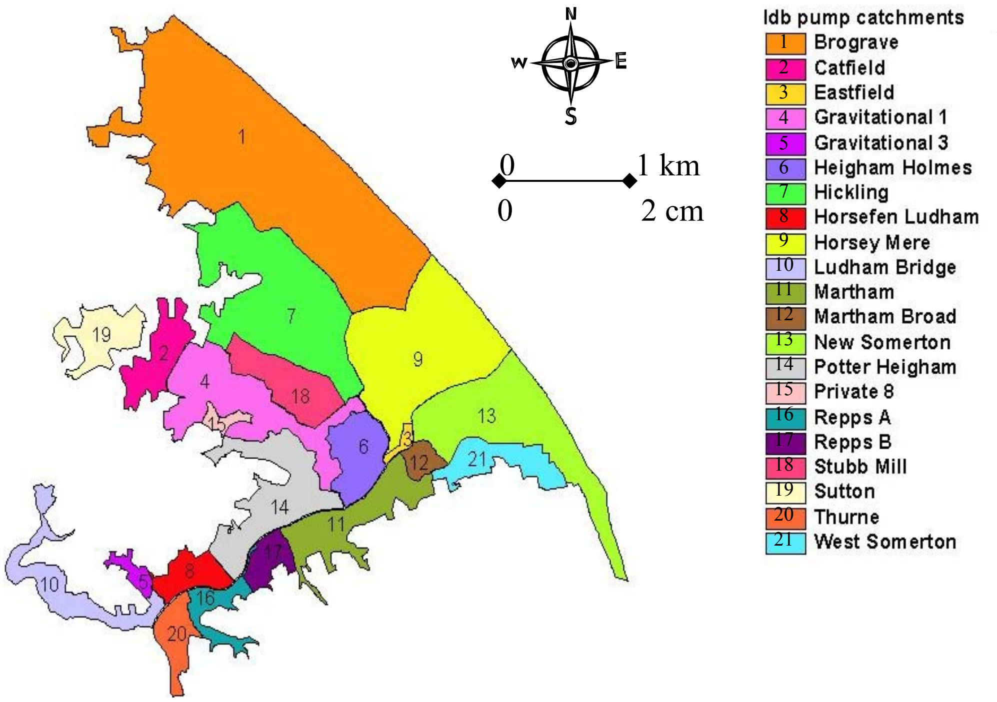


Figure 7.1.2: Upper Thurne IDB pump drainage areas

The land drainage pumps in the Upper Thurne river basin have therefore been modelled though the reservoir functions within SWAT. Reservoirs within SWAT are located on the main river channel network and receive water from all sub-basins upstream of the water body. They modify the movement of water in the channel network by lowering the peak flow. As the reservoirs slow down the flow of water, sediment will fall from suspension, removing nutrient and chemicals adsorbed to the soil particles. The volume of outflow from modelled reservoirs may be calculated using one of four different methods; measured daily outflow, measured monthly outflow, average annual release rate for uncontrolled reservoirs; controlled outflow with target release.

Sparse electrical consumption data from 1977 – 2001 are available for 5 pumps within the Upper Thurne watershed. From these data monthly mean discharge rates have been calculated as m^3 day^{-1} for the period from 1977 – 2001 using the following conversion factors for each pump:

Table 7.1.1: Upper Thurne Pump conversion factors (Holman, 1994)

Pump	**Conversion Factor m^3 kWh**
Catfield	153
Stubb Mill	83
Eastfield	66
Brograve	44
Horsey Mill	48

Each pump has therefore been modelled as a controlled outflow reservoir within SWAT. Calculated maximum and minimum average monthly discharge rates have been used to set monthly target release rates for each controlled reservoir. Figures 7.1.3, 7.1.4, 7.1.5, 7.1.6 and 7.1.8 show modelled pump rates. Although an exact flow pattern is not achieved due to actual pump rate variability a good comparison between annual pump rates has been seen (Table 7.1.2).

Table 7.1.2: Annual average observed and predicted pump flows in the Upper Thurne

Pump	Average annual pump flow (10^4 m^3 yr^{-1})	Modelled average annual pump flow (10^4 m^3 yr^{-1})
Catfield	520	470
Stubb Mill	953	978
Eastfield	2155	2310
Brograve	1572	1800
Horsey Mill	1711	1511

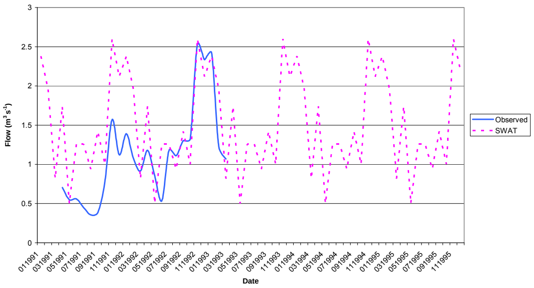


Figure 7.1.3: Observed and predicted pump comparison at Brograve

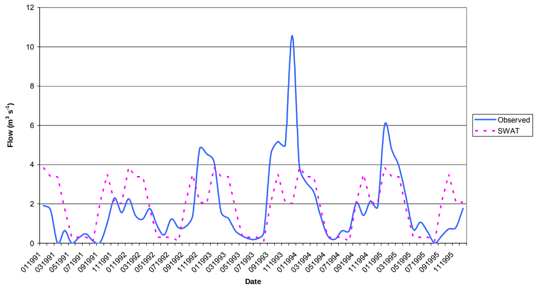


Figure 7.1.4: Observed and predicted pump comparison at Eastfield

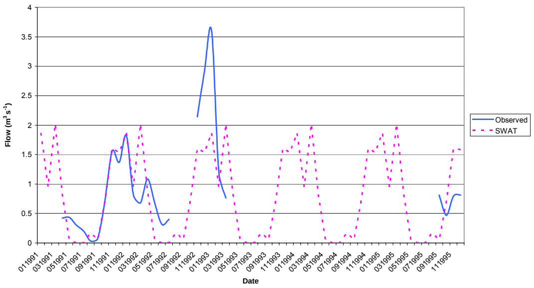


Figure 7.1.5: Observed and predicted pump comparison at Stubb Mill

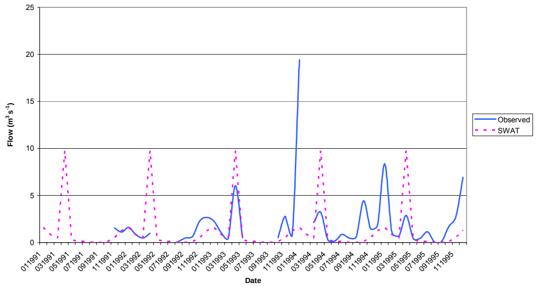


Figure 7.1.6: Observed and predicted pump comparison at Horsey

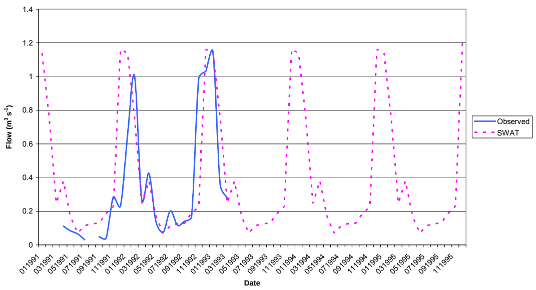


Figure 7.1.8: Observed and predicted pump comparison at Catfield

It can be seen from the above figures that SWAT is only able to represent Catfield pump with some degree of accuracy (Fig 7.1.8) matching both the timing and magnitude of peak flows reasonably well. It is thought that this may have occurred due to the delineation of the sub-basins within SWAT. SWAT delineates the river basin into topographic sub-basins (Fig 7.1.9). However, the sub-basins, which feed into the drainage pumps, are not topographic as they have been man-made and boundaries are placed around the edges of fields. Therefore the pump sub-basins, which have been modelled in SWAT, are different in size and shape to those of the actual pump sub-basins and are therefore affecting the amount of run-off each sub-basin will receive (Fig 7.1.9). The Catfield pump catchment is the only one, which is actually topographically defined due to the small size of the catchment. SWAT is therefore able to model reasonably well the flow pattern of this pump.

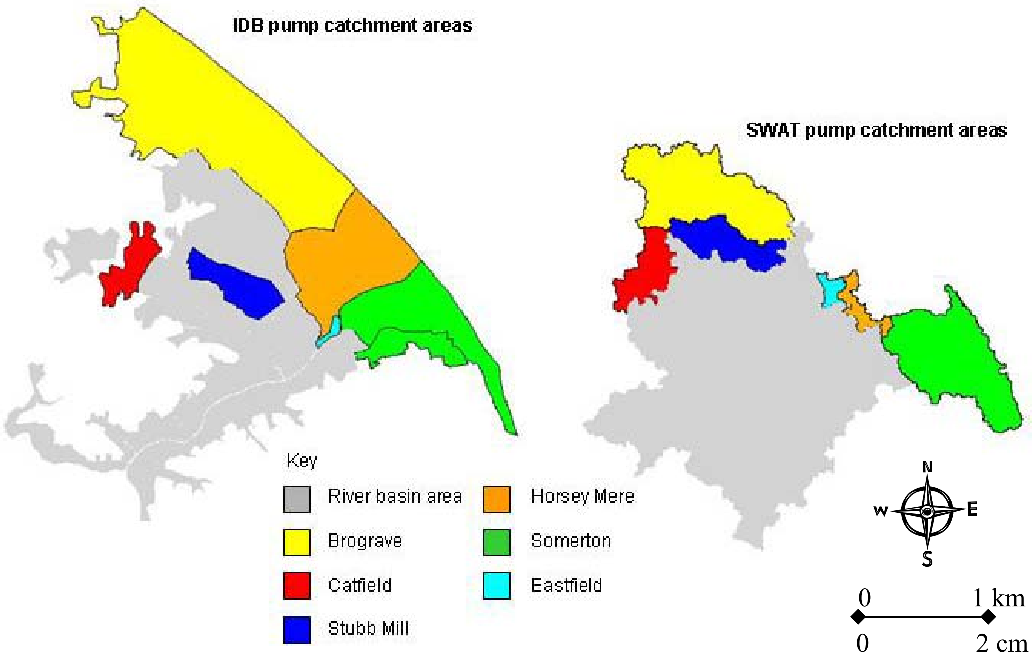


Figure 7.1.9: Comparison between IDB and SWAT pump catchment areas

7.2 Hickling Broad Modelling

Hickling Broad has been represented in SWAT as a pond file; a simple empirical model is used to predict the trophic status of the ponds. When calculating nutrient transformation in a pond file, SWAT assumes the system is completely mixed. In a completely mixed system, as nutrients and sediments enter the water body they are instantaneously distributed throughout the volume. The assumption of a completely mixed system ignores lake stratification and the intensification of phytoplankton in the epilimnion (upper waters of a thermally stratified lake subject to wind action).

Nutrient transformations simulated in ponds are limited to the removal of nutrients by settling. Transformations between nutrient pools (e.g. $NO_3 \leftrightarrow NO_2 \leftrightarrow NH_4$) are ignored. The settling rate in SWAT is input as m year^{-1}. For natural lakes, measured phosphorus settling velocities most frequently fall in the range of 5 to 20m year^{-1} although values over 200m year^{-1} have been reported (Chapra, 1997). Panuska and Robertson (1999) noted that the range in apparent settling velocity values for man made reservoirs tends to be significantly greater than for natural lakes. Higgins and Kim (1981) reported phosphorus apparent settling velocity values from -90 to 269m year^{-1} for 18 reservoirs in Tennessee with a median value of 42.2m year^{-1}. A negative settling rate indicates that the lake/reservoir sediments are a source of nutrients; a positive settling rate indicates that the reservoir sediments are a sink for nutrients.

Within the Broads region it has been widely reported that both the Broads and river sediments are sources of nutrients (Johnes 1996b, Moss *et al.* 1989, Stephen *et al.* 1997). Within the SWAT pond file the nutrient settling velocity for both phosphorus and nitrate has been set to -1m year^{-1} after recommended apparent settling velocity values (Table 7.2.1) from Panuska and Robertson (1999).

Table 7.2.1: Recommended settling velocity values for nutrients (Panuska and Robertson, 1999)

Nutrient Dynamics	**Range in settling velocity values (m year^{-1})**
Shallow water bodies with high net internal nutrient flux	$v \leq 0$
Water bodies with moderate net internal nutrient flux	$1 < v < 5$
Water bodies with minimal net internal nutrient flux	$5 < v < 16$
Water bodies with high net internal nutrient removal	$v > 16$

A number of other parameters can be input to SWAT within the pond file (Table 7.2.2). Data for these parameters have been predominantly obtained from long term averaged EA data for Hickling Broad. However the extent of eutrophication modelling within SWAT is limited to phosphorus inputs; this is due to the difficulty of controlling the exchange of nitrogen and carbon between the atmosphere and water and fixation of atmospheric nitrogen by some blue-green algae. This process over stimulates the growth of algae, causing unsightly scum and unpleasant odours, and robbing the water of dissolved oxygen vital to other aquatic life. Some blue-green algal species are also known to produce chemicals that can be toxic to wild and domestic animals, and also to man.

Reports show that the River Bure and its Broads are becoming dominated by blue-green algae during the summer months (Madgwick, 1999). It is thought that the demand for water from new commercial and housing developments in the catchments of the Broadland rivers has led to a reduction in the flow rates during the summer months. Consequently there has been a rise in the concentration of pollutants carried by these rivers, and a reduction in the rate at which broads associated with them are flushed by 'new' water derived from the catchment. The reduced flushing rate, together with the large nutrient load carried by the rivers, already results in blooms of blue green algae in some of the broads.

This would suggest that flow rates in the Broadland rivers are, from the ecological point-of-view, already less than they should be during the summer months, and historical evidence strongly supports this hypothesis (George, 1992). The effects of climate change will almost certainly exacerbate the problem of low fluvial flows in the rivers. In this respect, there is increasing evidence that this phenomenon will lead to summers in Eastern England becoming both warmer and drier over the coming decades. In the circumstances, it would seem that unless remedial action is taken, fluvial flow rates in the Broadland rivers during the summer could, in future be even lower than they are today. This in turn could have very damaging effects on the region's biodiversity and future well being. Unfortunately as future water demands are not known for the area and the fact that SWAT eutrophication modelling is limited to phosphorus inputs this phenomenon cannot be investigated further.

Table 7.2.2: Nutrient parameter values for SWAT

Parameter	Value	Source
Fraction of sub-basin area that drains into pond	0.3	Calculated by SWAT
Surface area of pond	141.1 ha	George (1992)
Volume of water stored in the pond	183.4 10^4 m^3	George (1992)
Initial sediment concentration	36.70 mg l^{-1}	EA average (1978 – 2005)
Phosphorus settling rate	-1 m yr^{-1}	Panuska and Robertson (1999)
Nitrogen settling rate	-1 m yr^{-1}	Panuska and Robertson (1999)
Chlorophyll *a* production	1.00 µg l^{-1}	SWAT default
Secci-disk depth	0.537 m	EA average (1978 – 2005)
Beginning month of mid –year nutrient settling season	February	Ascertained from EA average (1978 – 2005)
Ending month of mid-year settling season	July	Ascertained from EA average (1978 – 2005)

Figures 7.2.1 and 7.2.2 show SWAT predicted total phosphorus and total oxidised nitrogen values for Hickling Broad.

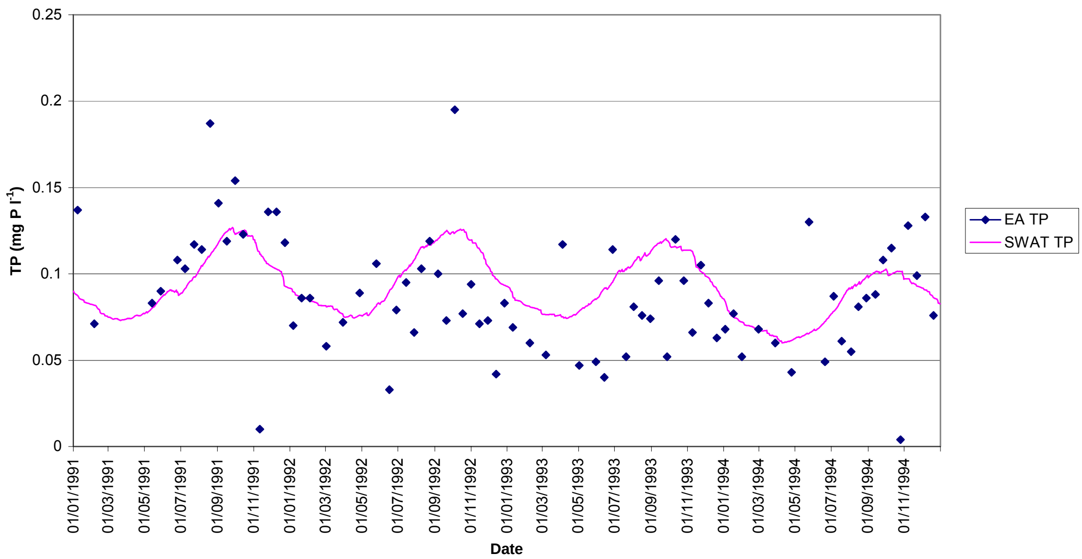


Figure 7.2.1: Observed and predicted total phosphorus in Hickling Broad (1991 – 1994)

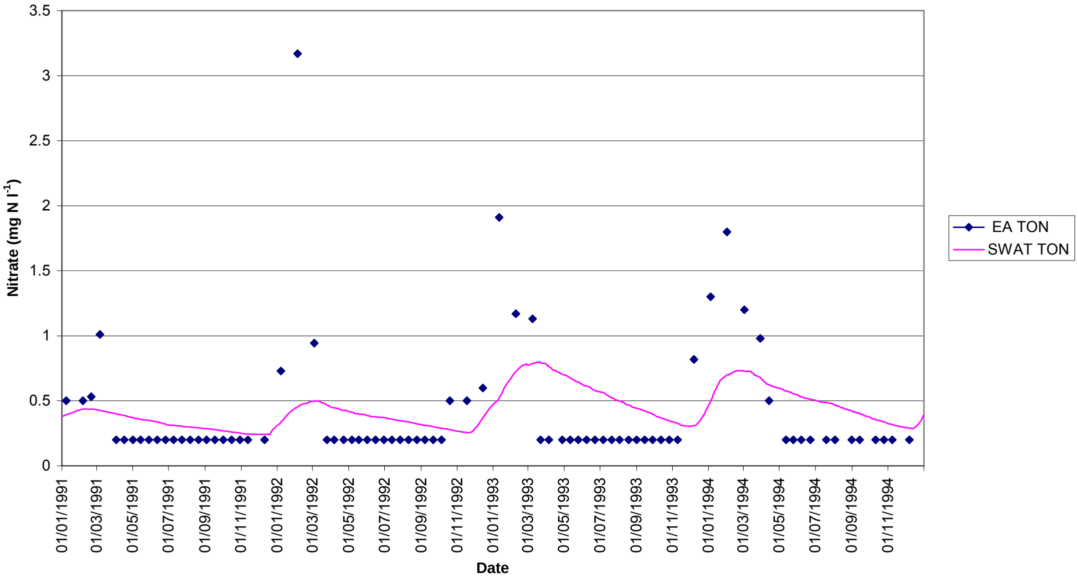


Figure 7.2.2: Observed and predicted nitrate values in Hickling Broad (1991 – 1994)

No long-term nitrate values are available for Hickling Broad. Therefore observed and modelled comparisons have been based on total oxidised nitrogen values. Long-term values for TON are available from the EA; NO_3 and NO_2 values have been summed from the SWAT output files to achieve comparable TON values.

From the TON graph it can be seen that SWAT does not predict the higher TON values and there are long periods of observed data showing constant TON values. This is because sample results are below confident detectable limits and have therefore been reported as ‘<’ values by the EA. Consequently an exact match between observed and modelled TON values cannot be obtained due to the limitations of the observed data.

7.3 Calibration and Validation

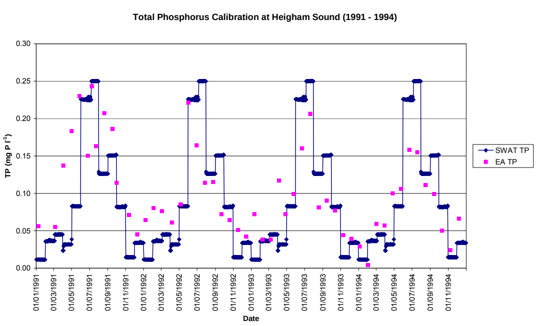
No observed flow or sediment data are available in the Upper Thurne, therefore calibration and validation has only been undertaken for nutrients. The EA takes water quality samples at three sites in the Upper Thurne catchment; unfortunately only two of these sites have suitable data for calibration purposes. These are Martham Ferry and Heigham Sound.

Calibrated land based nutrient parameters were taken from the Bure and Ant model and used within the Thurne model (section 7.0), none of these parameters were subsequently changed during the calibration procedure for the Thurne model. Instead calibration for these sites was done through changing the input parameters for the reservoir files further upstream as this controls the flow pattern of the Thurne model. Table 7.3.1 gives a summary of these parameters. Starting values were calculated from the limited observed nutrient data at each pump, where no values were available calibration started at 1.00.

Table 7.3.1: Reservoir parameters used for nutrient calibration parameters in the Thurne Model

Parameter (mg l^{-1})	Catfield		Eastfield		Horsey Mere		Somerton		Stubb Mill		Brograve	
	Start	Cal.	Start	Cal.	Start	Cal.	Start	Cal.	Start	Cal.	Start	Cal.
ORGP	0.149	0.143	0.037	0.037	0.096	0.088	0.58	0.64	0.048	0.048	0.08	0.096
SOLP	0.029	0.029	0.007	0.007	0.016	0.008	0.012	0.007	0.008	0.008	0.008	0.016
ORGN	1.00	1.00	1.00	0.00	1.00	0.00	1.00	1.00	0.00	0.00	1.00	0.00
NO_3	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	1.00	0.670
NH_3	0.3	0.30	1.00	1.00	1.00	1.00	0.38	1.00	0.504	0.504	0.344	0.344
NO_2	1.00	1.00	1.725	1.725	0.670	0.669	1.00	1.00	1.00	0.680	1.00	0.00

As with the Bure and Ant SWAT model, calibration was undertaken for the period 1991 – 1994. Figures 7.3.1 and 7.3.2 show the results of calibration at the two sites. As with Hickling Broad no observed nitrate data were available so calibration has been undertaken with TON values.



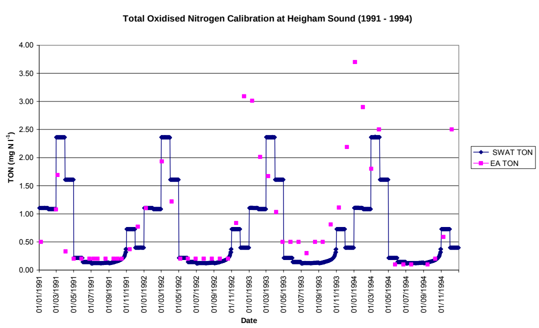


Figure 7.3.1: Nutrient calibration at Heigham Sound

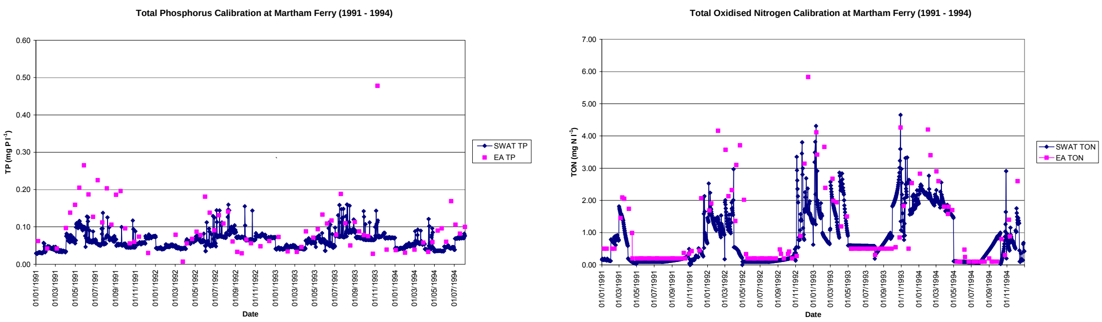


Figure 7.3.2: Nutrient calibration at Martham Ferry

Calibration results show that there is no variation between individual year's nutrient patterns, especially at Heigham Sound. This is further emphasised with validation results (Fig 7.3.3 and 7.3.4). At Heigham Sound there is no daily variation of nutrient concentrations, this can also be seen for TP concentrations at Martham Ferry but not to the same extent. It is thought that this is a result of the SWAT modelling of the pumps in the Upper Thurne river basin.

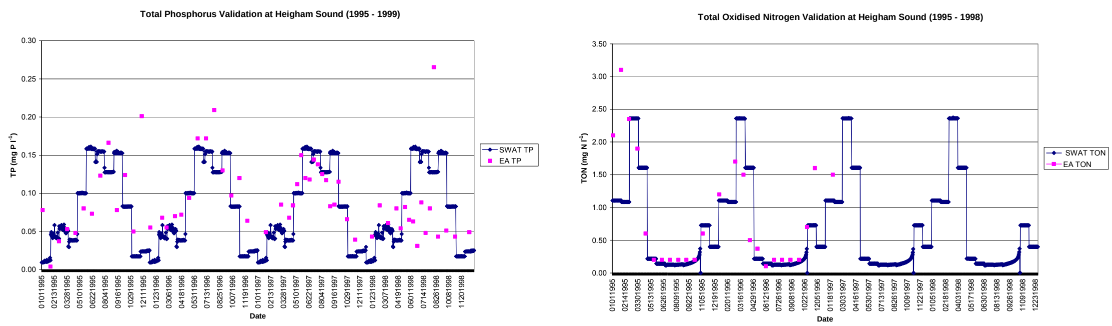


Figure 7.3.3: Nutrient validation at Heigham Sound

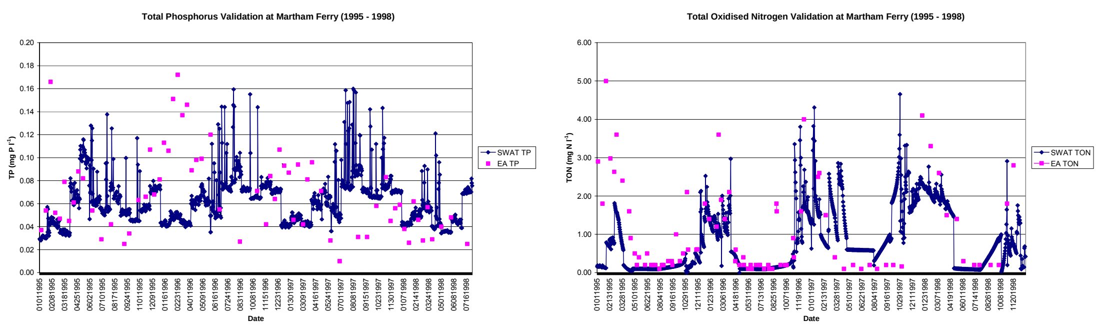


Figure 7.3.4: Nutrient validation at Martham Ferry

Pump discharge obviously influences the volume of water in the river and dyke system of the Upper Thurne. As only monthly discharge data were available, the reservoir files used to characterise the pumps within SWAT have only been set up to simulate monthly target release rates. Therefore there is no daily variation of pump discharge; the pumps will discharge the same volume of water for each day within a set month.

However, both the calibration and validation graphs show that SWAT is able to simulate nutrient concentrations with some degree of accuracy. As discussed before these data are misleading as observed data have only been recorded on a monthly or bi-monthly basis, and are consequently not representing the daily variation of nutrient concentrations within the system. Due to this, the modelling of the pumps within the Upper Thurne system and the lack of flow data to calibrate SWAT, the model can not be used to simulate results at a daily level with any degree of confidence. The Upper Thurne SWAT model should therefore only be used to simulate annual or monthly results when looking at future scenarios.

Chapter Eight **Future Scenarios**

8.0 Future Scenarios

Scenarios are neither predictions nor forecasts of future conditions. Rather they describe alternative plausible futures that conform to sets of circumstances or constraints within which they occur (Hammond, 1996). The true purpose of scenarios is to illuminate uncertainty, as they help in determining the plausible futures (Fisher, 1996). The construction of scenarios will include climate, socio-economic and land use processes.

As previously discussed in chapter two, climate changes affect the hydrological cycle, thus modifying the transformation and transport characteristics of nutrients. At the current stage of knowledge, large-scale global circulation models (GCM) are probably the best available tools to estimate these changes (Bouraoui *et al*., 2002). Within the UK the UK Climate Impacts Programme (UKCIP) provides scenarios that show how climate might change and co-ordinates research on dealing with the future climate. The UKCIP02 climate change scenarios present four different descriptions of how climate may change based on four different emission scenarios. The scenarios provide alternative views of the future, and together show a broad range of changes that may occur in the future. The scenarios were commissioned and funded by the Department of Environment, Food and Rural Affairs (DEFRA) for UKCIP, and developed by the Tyndall Centre for Climate Change Research at the University of East Anglia and the Hadley Centre for Climate Prediction and Research at the Meteorological Office.

The scenarios have been developed using the latest global climate model from the Hadley Centre for Climate Prediction and Research at the Meteorological Office, HadCM3 is a coupled atmosphere-ocean general circulation model (AOGCM) developed at the Hadley Centre and described by Gordon *et al*., (2000) and Pope *et al*., (2000). Unlike earlier AOGCMs at the Hadley Centre and elsewhere (including HadCM2), HadCM3 does not need flux adjustment (additional "artificial" heat and freshwater fluxes at the ocean surface) to produce a good simulation. The higher ocean resolution of HadCM3 is a major factor in this. HadCM3 has been run for over a thousand years, showing little drift in its surface climate.

To model future climate and land use scenarios in SWAT a number of parameters will have to be changed. To simulate possible future climate scenarios new weather files will need to be created. This will include changing rainfall, temperature, relative humidity, wind, potential evapotranspiration and radiation inputs to the model. This will allow the impact of climate change on the hydrology of the system to be assessed, and to see how these changes will affect nutrient transport and loading on the system.

In order to model future land use change a further set of parameters will need to be adjusted. These all fall within the management files of the SWAT model and include type of crop, area of crop, crop rotations, amount and timing of irrigation as well as crop sowing and harvesting dates. Management files need to be changed not only to represent future changes in land use but also to represent the impact future climate might have on crop growth and how changes in irrigation might affect the soil moisture.

8.1 Climate Change and Land Use Scenarios

Future socio-economic and land use scenarios by Morris (2003) together with the RegIS project in East Anglia, which also includes future climate scenarios, have been reviewed. RegIS was developed as an impact study for the UK climate Impact Programme (Holman and Rounsevell, 2001). The principal aim of RegIS was the development of a robust and transparent methodology for stakeholder-led, regional assessment of climate change impacts and cross-sectoral interactions between the major sectors driving landscape change. The methodology was developed in the North West and East Anglia, and is believed to be transportable to other regions of the UK, thereby providing a framework for further assessments and studies. The RegIS project represents the first attempt at quantitatively modelling the cross-sectoral impacts of climate change within an integrated framework at a regional scale within the UK.

The methodological framework for this study was funded by DETR. In RegIS, the UKCIP02 climate scenarios have been used, and the national socio-economic scenarios provided by UKCIP have been developed to provide the regional socio-economic data necessary for the integrated RegIS methodology. UKCIP also provided resources and expertise to access the other extensive data sets required by the study and to transform them into the formats needed to run the models on a regional framework. It was a basic objective of the programme to enable use of common data sets. Therefore, it is proposed to use these known and approved data sets which are easily accessible within the SWAT modelling as future scenarios, as it combines known future climate scenarios with future socio-economic scenarios, covering the study area within one of its test regions; East Anglia. Future climate and socio economic scenarios used within RegIS can be seen in Table 8.1.1.

Table 8.1.1: RegIS scenarios

Climate Scenario	Pressure on Environment	Socio – Economic Scenario	Pressure on Environment
UKCIP02 2050s high	Temperature increases by 2.3 ^{o}C. Rainfall increases by 2%. Rainfall increases can be seen in the winter, autumn and spring but a decrease in summer rainfall is predicted. PET increases by 29%.	Regional Enterprise (RE)	Highest socio-economic pressure – an extreme case of a society that does not respond to the threat of climate change over the next 50 years i.e. an 'adverse case' analysis.
UKCIP02 2050s low	Temperature increases by 0.9 ^{o}C. Rainfall increases by 1%. Rainfall increases can be seen in the winter, autumn and spring but a decrease in summer. PET increases by 14%	Global Sustainability (GS)	Lowest pressure - 'better case' analysis with respect to pressures upon environmental systems and associated impacts.

Within RegIS the UKCIP02 high and low climate scenarios are from output by the HadCM3 GCM (Hulme and Jenkins, 1998). Changes in mean annual temperature of +0.9 and +2.3 °C are projected for the Low and High scenarios for the 2050s, respectively in East Anglia. Seasonal changes in temperature are similar to the annual projections. Annual precipitation changes of +1 and +2 % are projected. Seasonal differences in precipitation changes are much greater than for temperature, with increases generally projected in winter, autumn and spring and decreases in summer. Annual potential evapotranspiration increases by +14 and +29 % in East Anglia.

The Regional Enterprise socio- economic scenario is characterised by the emphasis on private consumption but with decisions made at national and regional level. This will reflect local priorities and interests. Market values will dominate, with crops being produced for the domestic market, primarily through supermarkets. Global sustainability is characterised by more pronounced social and ecological values, which are evident in global institutions and trading systems. There is collective action to address social and environmental issues. Growth is slower but more equitably distributed compared with other socio-economic scenarios.

The high climate change scenario combined with the Regional Enterprise (RE) socio-economic scenario is likely to impose the highest socio-economic pressure upon the agricultural sector. As the socio-economic scenarios contain no element of the climate change scenarios, this provides an extreme case of a society that does not respond to the threat of climate change over the next 50 years i.e. an ‘adverse case’ scenario (Holman and Rounsevell, 2001). The lowest climate change scenario is combined with the Global Sustainability (GS) socio-economic scenario which brings with it the lowest pressure upon the agricultural sector i.e. a ‘better case’ analysis with respect to pressures upon environmental systems and associated impacts.

8.1.1 Available Data

Future climate data and land use data have been derived from three sources. The RegIS project has provided the future land use data. Land use data were provided for 5 km^2 grid squares, within each grid square the cropping area in hectares for each crop was given for each future scenario (Table 8.1.2). These data were utilised to provide crop types and cropping area for the future scenarios within the SWAT management files. Comparisons between future cropping areas and current modelled SWAT cropping areas are discussed in section 8.2.

Table 8.1.2: Example of selected RegIS grid squares for use in SWAT management files (high RE scenario)

Easting	Northing	Forest (ha)	Winter Wheat (ha)	Winter Barley (ha)	Spring Barley (ha)	Oats (ha)	Potatoes (ha)	Sugar Beet (ha)
RegIS grid cells for the Bure and Ant SWAT model (high RE scenario)								
610000	335000	411	216	396	197	219	25	336
610000	330000	154	242	412	185	195	48	360
605000	330000	221	261	416	182	207	42	366
605000	325000	20	477	594	237	275	39	527
615000	330000	79	290	470	197	214	63	411
610000	325000	124	286	490	214	230	55	425
630000	325000	207	241	338	122	145	47	298
630000	330000	126	217	330	121	137	45	283
625000	325000	269	253	413	169	191	45	358
630000	320000	118	292	431	153	180	51	369
615000	320000	459	190	350	160	182	23	293
620000	320000	116	269	420	168	199	46	363
630000	315000	291	87	133	50	56	16	114
620000	315000	272	133	253	118	135	11	209
635000	320000	188	82	115	41	43	18	100
630000	310000	152	277	444	171	200	43	375
635000	315000	95	223	266	79	92	33	197
615000	325000	469	207	346	153	163	40	303
625000	330000	49	246	420	183	197	50	365
625000	320000	57	306	490	186	226	52	420
615000	335000	152	264	450	199	211	53	393
620000	325000	30	319	505	201	238	57	440
620000	330000	181	306	507	215	231	64	441
RegIS grid cells for the Thurne SWAT model (high RE scenario)								
630000	310000	152	277	444	171	200	43	375
630000	315000	291	87	133	50	56	16	114
630000	320000	118	292	431	153	180	51	369
630000	325000	207	241	338	122	145	47	298
635000	310000	121	347	476	160	190	53	376
635000	315000	95	223	266	79	92	33	197
635000	320000	188	82	115	41	43	18	100
635000	325000	29	256	360	132	139	59	315
640000	310000	28	269	291	74	93	38	200
640000	315000	21	157	183	52	64	24	136
640000	320000	22	7	7	2	2	2	5
640000	325000	34	1	1	0	0	0	0
645000	310000	55	333	404	118	145	50	308
645000	315000	80	265	366	115	153	43	293
645000	320000	32	0	0	0	0	0	0
650000	310000	1	139	165	41	63	19	119
650000	315000	0	27	39	13	16	5	33

Daily weather data were provided by the MONARCH (Modelling Natural Resource Responses to Climate Change) data series. The data are from the ADAS daily climate scenario dataset developed using the LARS model and the UKCIP02 scenarios for the DEFRA funded cc0368 project. The principal aim of the MONARCH study was to evaluate the direct impacts of climate change on the natural conservation resources of Britain and Ireland through an integrated methodology linking established impact models to a broad-scale bioclimatic classification (Harrison *et al*, 2001).

The data series consists of daily baseline (1961 – 1999) and climate change scenarios on a 5 km^2 grid with UKCIP02 and HadCM3 projections for 2020 and 2050 Low and High emissions. These are based on the LARS-WG stochastic weather generator (Semenov et al., 1998). LARS-WG provides daily data on maximum and minimum temperature, precipitation, radiation and potential evaporation. For each weather variable LARS-WG specifies daily probability distribution and statistical relationships between the variables. The data provides 50 years of baseline and climate change scenarios. However data were generated independently for each grid and there is no correspondence between years in neighbouring grids. Therefore climate data from the MONARCH grid cells, which contained the ten BADC rain gauges, used in the model build process (Chapter Five) were abstracted (Table 8.1.3).

Table 8.1.3: MONARCH grid cells corresponding with BADC rain gauges used within the SWAT model

Rain Gauge	MONARCH Grid Cell
Aylsham Bankfield House	E6175N3275
Barton	E6375N3225
Coltishall	E6275N3375
Hemsby	E6475N3175
Hickling	E6425N3225
Hindolveston Hope House	E6025N3275
Melton Constable	E6025N3325
South Repps	E6275N3375
Wolterton Park	E6175N3325
Woodgate House	E6175N3275

For baseline and future conditions rainfall ratios between the cells containing rain gauges and the cell containing Aylsham Bankfield House gauge were calculated (Table 8.1.4). The Aylsham Bankfield House gauge was used as it was shown to have rainfall closest to the areal average for the study area as discussed in Chapter Five. The ratios were then used to scale daily rainfall for each grid cell used. Ratios were not calculated for other climate parameters such as temperature or potential evapotranspiration as only one gauge was available for the study area and utilised in the SWAT model (Coltishall). Figure 8.1.1 shows there is a good comparison between current calculated PET values and MONARCH baseline PET values, with both sets of data reaching the same peak PET value of 121 mm.

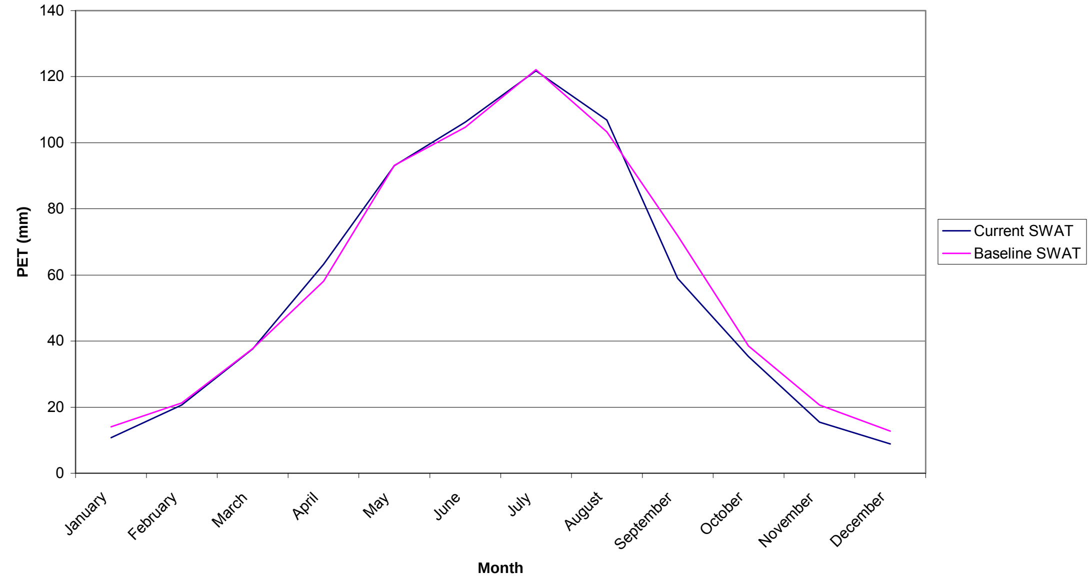


Figure 8.1.1: Comparison between current modelled PET (Hargreaves method) and MONARCH baseline PET

Table 8.1.4: Calculated rainfall ratios from MONARCH data series for baseline and future scenarios within SWAT

		Rain Gauge Ratio								
Month	Aylsham monthly total (mm)	Barton	Coltishall	Hemsby	Hickling	Hindolveston	Melton	South Repps	Wolterton	Woodgate House
January	52.10	0.98	1.02	0.98	0.96	0.82	0.85	1.14	0.93	1.00
February	39.29	1.03	0.97	0.95	0.98	0.90	0.88	1.10	0.96	1.00
March	42.86	0.99	1.05	1.01	0.90	0.92	0.80	1.01	0.89	1.00
April	47.20	0.96	1.07	1.07	0.96	0.83	0.93	1.04	1.02	1.00
May	46.91	1.06	1.10	1.07	1.00	0.84	0.88	1.03	0.89	1.00
June	48.72	1.07	0.99	1.03	1.16	0.86	0.88	0.95	1.00	1.00
July	46.48	0.90	0.90	0.91	0.96	0.81	0.65	0.98	0.80	1.00
August	62.12	1.10	1.06	1.27	0.94	1.02	0.95	1.33	1.24	1.00
September	49.70	1.01	1.14	1.14	0.90	0.91	0.81	1.00	0.99	1.00
October	52.68	1.04	0.94	0.93	1.03	0.89	0.81	0.88	0.97	1.00
November	66.83	1.02	1.10	0.92	1.15	0.98	0.89	1.12	1.12	1.00
December	54.74	0.94	1.01	1.00	1.02	0.93	0.96	1.01	0.93	1.00

The two original SWAT models for the study area were based on observed climate data for the calibration and validation period. It was however considered more appropriate to use the daily baseline data generated by MONARCH for comparisons between current and future scenarios to be made. This would ensure that comparisons were made between sets of modelled climate data rather then between observed and modelled data and therefore any errors would be consistent. Model runs were undertaken using the baseline data for the calibration and validation period and compared to results from runs with observed climate data. Figure 8.1.2 shows that the baseline rainfall is similar to observed BADC rainfall, although there is a tendency to over predict. Figure 8.1.3 mirrors the rainfall data showing that although the flows are similar the MONARCH rainfall data predicts higher flows. These are still however comparable to observed flows at Horstead Mill, this can be seen in Table 8.1.5 where there is no significant difference between the baseline and current modelled means flows at Horstead Mill when using the student *t* test.

Table 8.1.5: Results of a Student *t* test comparing the difference between baseline and current modelled mean flows at Horstead Mill

Statistic	Baseline	Current
Mean	5.77	4.43
Standard deviation	41.23	3.87
Variance	32.12	15.01
Observations	15	11
Degrees of freedom	24	
T_{cal}	0.12	
T_{tab}	2.064	
Significance at 5% level	NO	

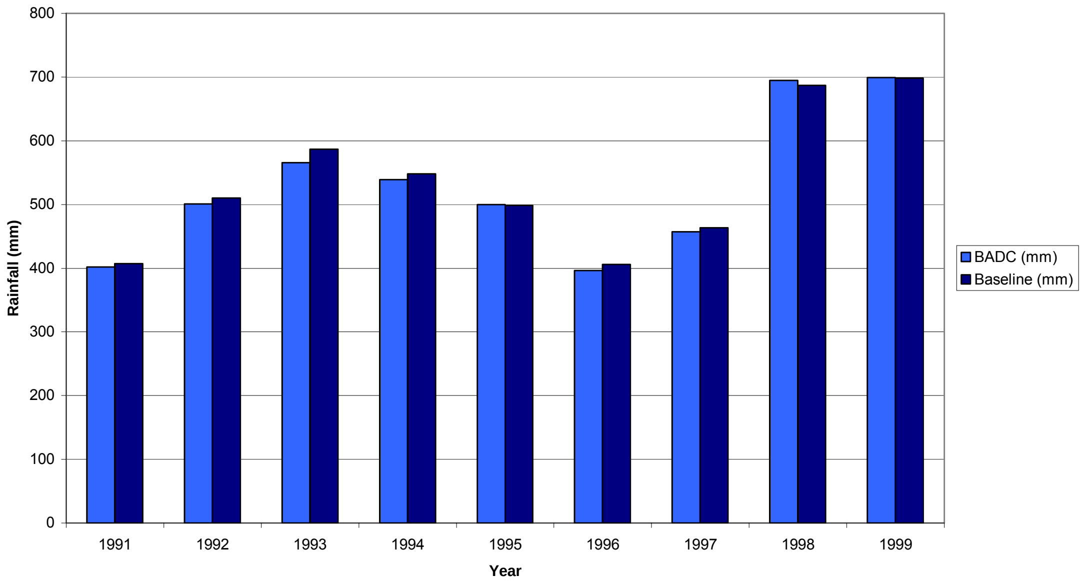


Figure 8.1.2: Comparison between BADC rainfall (Aylsham) and MONARCH Baseline data (E6175N3275)

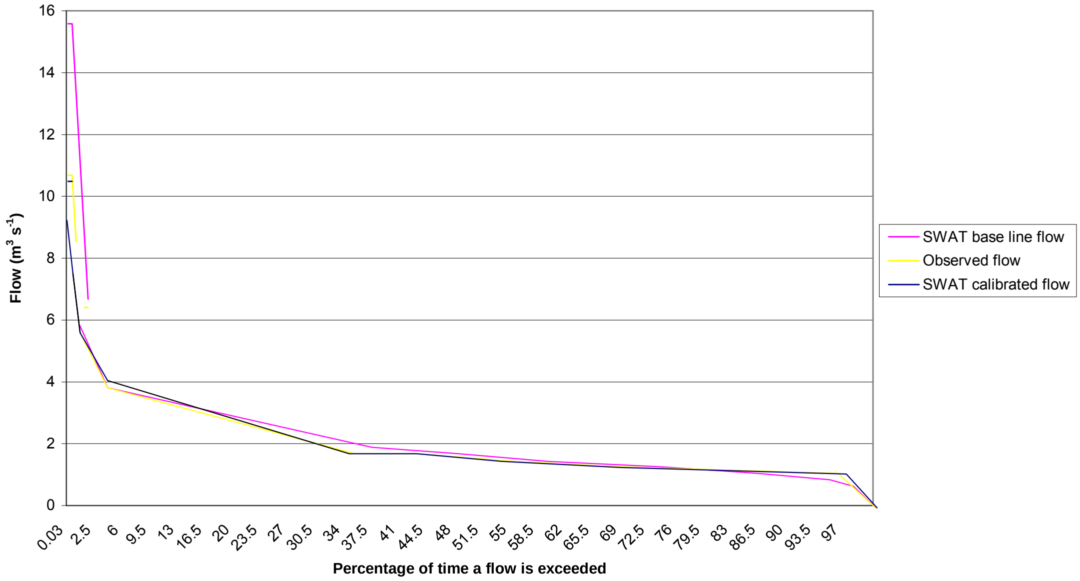


Figure 8.1.3: Flow duration comparison between model runs with SWAT BADC climate data and MONARCH baseline climate data for Horstead Mill (1991 – 1999)

Crop planting and harvesting dates were provided by the ACCELERATES project. The ACCELERATES project seeks to examine the relationship between agricultural land use responses to environmental change drivers and environmental protection. It is funded by the Directorate General for Research of the Commission of the European Communities, within the Fifth Framework Programme under the Energy, Environment and Sustainable Development Sub-Programme and is available at http://www.geo.ucl.ac.be/accelerates/. These data were incorporated into SWAT as the increased temperatures from the 2050 high and low climate scenarios will allow crops to be sown at earlier dates and will allow crops to develop faster and therefore be harvested earlier. Autumn sowing of winter crops such as winter wheat and barley will be delayed because of too much autumn growth and therefore the potential increase of frost damage over the winter months. Figure 8.1.4 shows that climate change has very little effect on the planned number of workable days as, although evaporation and rainfall are greater, large rainfall events, which change the soil state from workable to non-workable, occur on the same days in the baseline and climate change scenarios (Audsley *et al*., 2001).

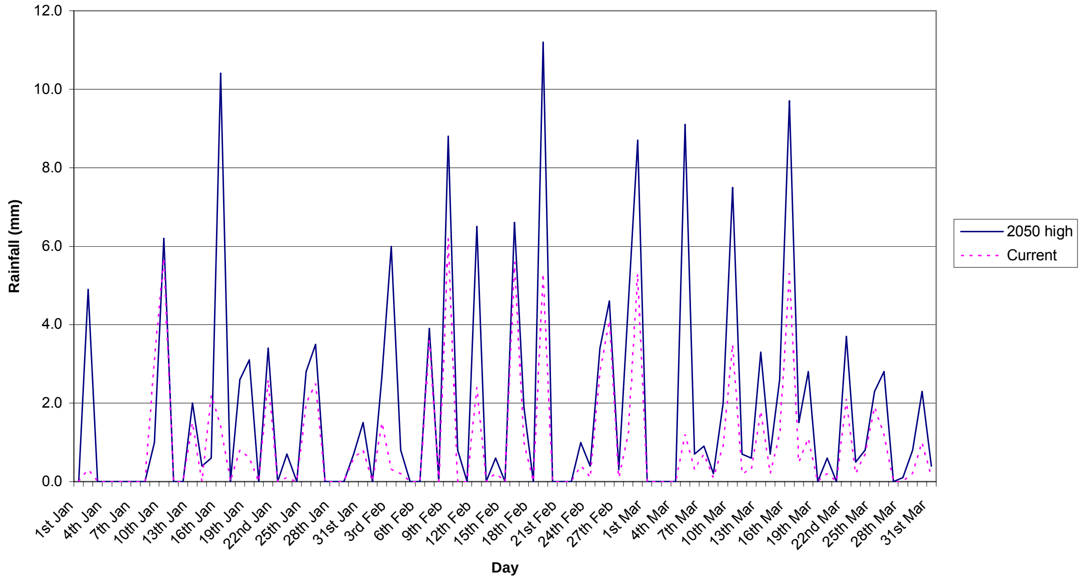


Figure 8.1.4: The occurrence of wet days for current and 2050 high rainfall at Aylsham Bankfield House

8.2 Changes to SWAT model

All the changes to the calibrated and validated SWAT model occurred in the weather and management files. New weather files were created for the Baseline, 2050 low and 2050 high scenarios. Available data from the MONARCH data series only consist of daily minimum and maximum temperatures, daily precipitation, daily solar radiation and potential evapotranspiration. Relative humidity and wind speed for the baseline and future scenarios were not included in the model as SWAT only requires these parameters if the Penman-Monteith equation is going to be used to calculate evapotranspiration. As evapotranspiration values are provided for baseline and future scenarios in the MONARCH data series evapotranspiration values were once again read into the SWAT model.

Within the management files new crop rotations were created based on the RegIS data for the 2050 high RE and the 2050 low GS scenarios. The same method (as described in Chapter Five) was utilised to distribute the crops within the study area. Figure 8.2.1 shows the area covered by each crop for the Bure and Ant SWAT model as well as the Thurne model. It can be seen from the graphs that a greater variety of crops are represented in the 2050 low GS scenario, however for both sites crop patterns are similar for each scenario. The 2050 low GS scenario is dominated by winter wheat, in comparison the 2050 high RE scenario is dominated by winter barley. Less forested area is present in the 2050 high RE scenario for both areas, within both the future scenarios the overall forested area has increased compared to EDL 2000 data. There is also an increased amount of potatoes and the introduction of oats in both the future scenarios. A reduction of winter crops can be seen between the future 2050 scenarios and the 2000 EDL data. This is due to the corresponding increase of spring crops such as sugar beet and potatoes in the future scenarios.

Neither oats nor linseed is present in the original 2000 EDL data used in the calibrated and validated SWAT model. There is also the noticeable omission of set-aside in the two future scenarios. Within the EDL 2000 data both modelled areas have a considerable amount of set aside; however set aside only occurs in the Bure and Ant 2050 Low GS scenario.

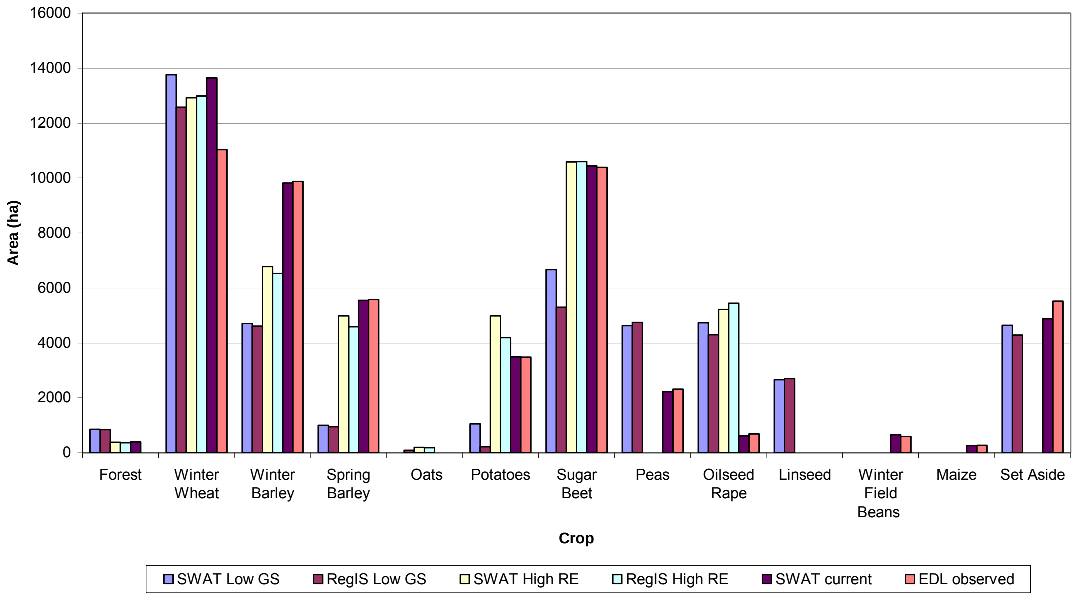


Figure 8.2.1: Crop areas for the 2050 High RE and 2050 Low GS scenarios taken from RegIS data compared to current EDL crop areas utilised in the SWAT model for the Bure and Ant catchments

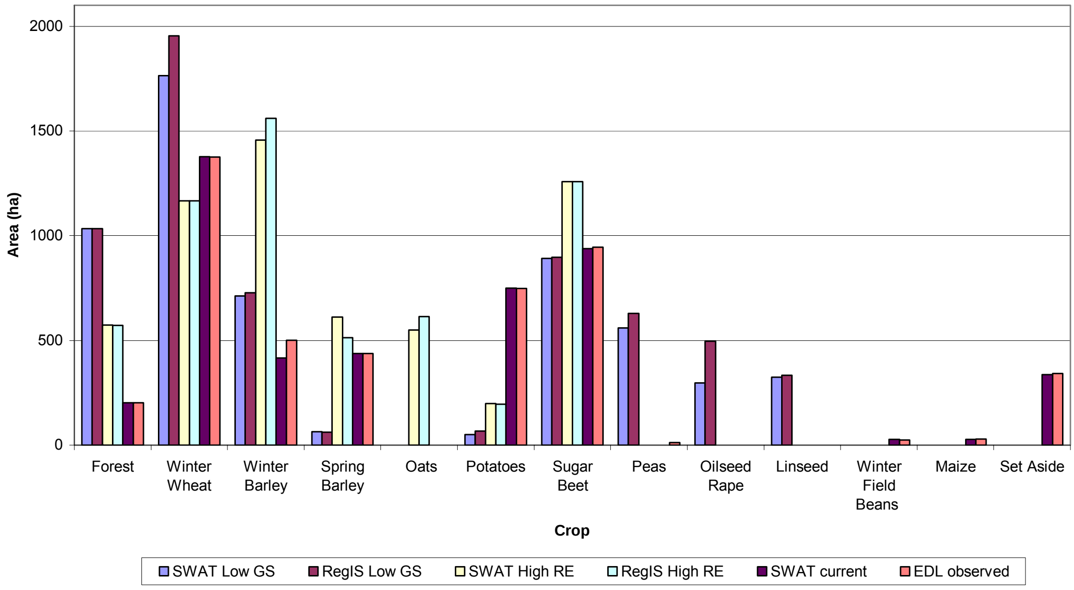


Figure 8.2.2: Crop areas for the 2050 High RE and 2050 Low GS scenarios taken from RegIS data compared to current EDL crop areas utilised in the SWAT model for the Thurne catchment

Rotations were based on those previously created for the calibration and validation of the SWAT model, which were originally derived from ADAS standard crop rotations as described in Chapter Five. As there is an increased amount of sugar beet and potatoes and the inclusion of oats and linseed the rotations had to be adjusted accordingly. This was achieved by replacing rotations originally including peas and winter field beans to include sugar beet and potatoes instead. Where rotations were originally adjusted from 'other mineral' to 'sandy' ADAS rotations to reduce the amount of winter wheat (Chapter Five), these re-adjusted back to 'other mineral' rotations to account for the decreased variation of crops within the future scenarios.

It was not feasible to include every possible crop that can be grown in the UK and that might be grown in the future. The major crops were selected in order to provide, as far as possible, the full range of crop types. Crop planting and harvesting dates were taken from the ACCELERATES project which provided the latest possible sowing dates for each crop and harvest dates which were set to be a fortnight after maturity Table 8.2.1 shows the original planting and harvesting dates compared to the new future scenario dates.

Table 8.2.1: Crop planting and harvest dates for future scenarios

	Original SWAT model		2050 Low GS		2050 High RE	
Crop	**Sow**	**Harvest**	**Sow**	**Harvest**	**Sow**	**Harvest**
Winter Wheat	15th Oct	20th Aug	10th Dec	4th Aug	9th Dec	2nd Aug
Maize	24th Apr	26th Oct	6th May	29th Sept	6th May	26th Sept
Oilseed Rape	31st Aug	30th Jul	14th Sept	7th Jul	14th Sept	6th Jul
Potatoes	1st Apr	13th Sept	4th Apr	14th Sept	3rd Apr	13th Sept
Winter Barley	1st Oct	31st Jul	2nd Dec	29th Jul	30th Nov	28th Jul
Spring Barley	20th Feb	8th Aug	23rd Jan	16th Jul	14th Jan	10th Jul

Figures 8.2.3 and 8.2.4 show vegetation growth for the dominant crops under the low GS and high RE future scenarios. It can be seen that published peak LAI are reported by Hough (1990) to lie between 3-8 for all the above crops, all of the modelled crops fall within this range. Leaf area index (LAI) decreases once plants reach senescence, and biomass falls somewhat later after maturity, approximately 2 weeks before harvest. It can therefore be seen that all plants are growing in the expected manner under the future scenarios.

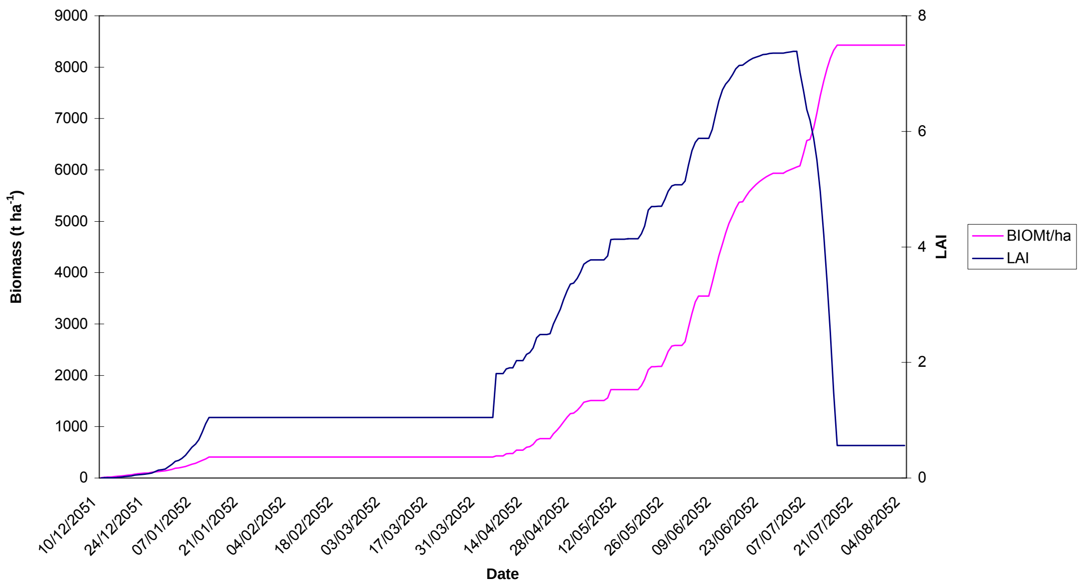


Figure 8.2.3: Winter Wheat growth for the 2050 low GS scenario

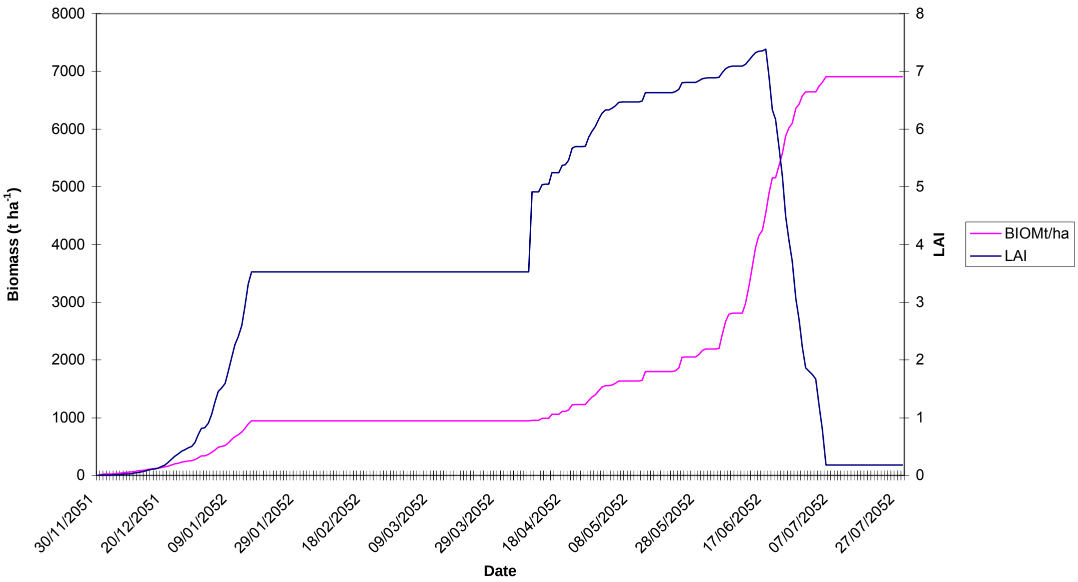


Figure 8.2.4: Winter barley growth for the 2050 high RE scenario

Using climatic data from the MONARCH data series CropWat was used to produce irrigation schedules for each irrigated crop. These were then transferred to the SWAT management files. Consequently irrigation timings and amounts were changed within the model. It can be seen from Table 8.2.2 that as expected there is an increase in irrigation volumes for all the crops with the future scenarios except spring barley where there is a decrease in irrigation volume. This is due to the earlier harvest date, as with the original SWAT model the majority of the total irrigation volume was applied in the month of July.

Table 8.2.2: CropWat irrigation amounts for future scenarios and use within SWAT based on light soils

Crop	Original SWAT model total annual average irrigation (mm yr^{-1})	2050 Low GS total annual average irrigation (mm yr^{-1})	2050 High RE total annual average irrigation (mm yr^{-1})
Sugar Beet	373.5	691.7	732.4
Maize	250.7	464.0	475.8
Spring Barley	152.7	67.7	70.2
Winter Wheat	100.0	525.6	539.0
Potatoes	308.3	566.5	601.4

Water for irrigation in England and Wales constitutes a small proportion of total abstraction nationally, but it is a consumptive use, peaking in summer months in dry years when water resources are scarcest. In many catchments the volume of water licensed for abstraction is now considered environmentally unsustainable (Knox *et al*., 2000 & EA, 2001). The high irrigation volumes predicted by CropWat under the future scenarios (Table 8.2.2) coupled with the low summer flows seen in Fig 8.3.1 suggest that current agricultural practices are not sustainable under future scenario conditions. Therefore future scenario modelling within SWAT has been undertaken assuming unlimited water supplies although it is recognised that this not sustainable in the future.

8.2.1 Model Runs

To assess the impact of future climate and socio-economic scenarios upon the Bure, Ant and Thurne watersheds a total of 5 model runs have been undertaken for each modelled river basin (Table 8.2.3). Management files from the calibrated and validated SWAT model were run with the baseline weather files to compare results between the baseline climate and observed climate data for the period 1990 – 1999. This was to ensure that the baseline climate data represented actual observed data with some degree of accuracy. These original management files were also run with the two future climate scenarios to assess the impact of climate change upon the study area. The last two runs incorporated both future land use and climate to assess the effect of combined land use and climate change on the study area.

Table 8.2.3: Future scenario model runs for SWAT

Land Use/Climate Change	Baseline	2050 high	2050 low
Original model	✔	✔	✔
Global Sustainability			✔
Regional Enterprise		✔	

A model run time of 20 years was calculated based on the hydraulic conductivity of the soil. The movement of water was calculated based on a K_{sat} of 3m day^{-1} for a sandy loam/fine sand soil as suggested by Smedema and Rycroft (1988) and a maximum watershed distance of 21048.61m based on the largest sub-basin within the

SWAT model, the distance which soil water will have to travel before reaching the watercourse. Therefore it would take 7016.2 days for groundwater to reach the watercourse, which is approximately 20 years. By running the model for this time period it will allow the affect of recharge and nutrient loads from groundwater to be assessed.

8.3 Results of Future Climate Model Runs (no land use change)

As discussed in Chapter Six only monthly or annual SWAT flows could reliably be looked at for future scenarios due to the problems of modelling the land drainage pumps within the Thurne river basin. Figure 8.3.1 shows daily flows within the Bure and Ant model for the baseline and 2050's high and low climate scenarios.

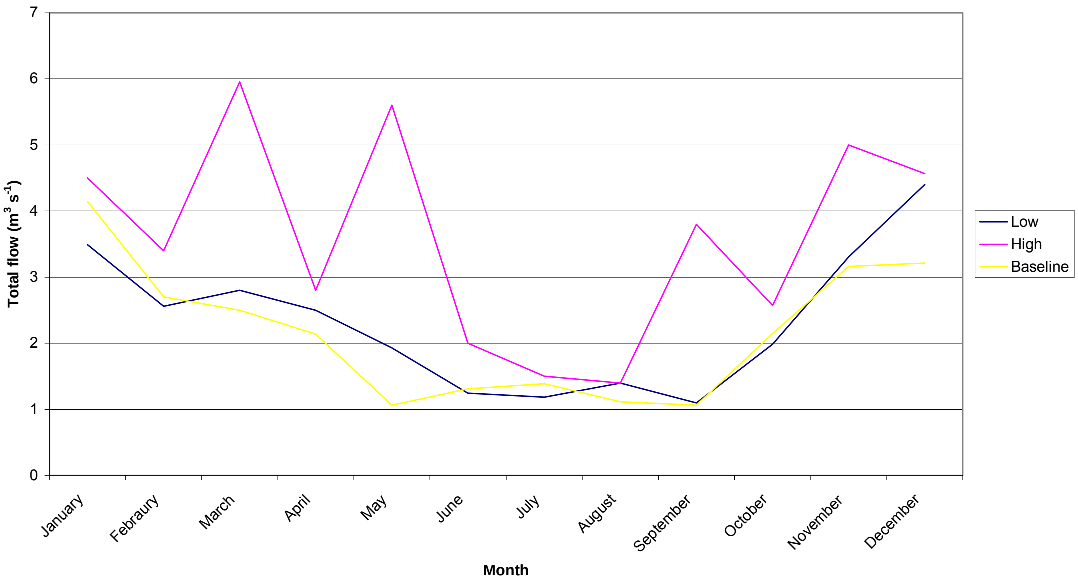


Figure 8.3.1: SWAT daily flows (Bure and Ant) for the baseline and 2050's high and low climate scenarios

As can be seen from the above graph higher winter flows can be expected for both the future climate scenarios. The 2050's low scenario follows a similar annual pattern to that of the baseline scenario. The 2050's high scenario is very peaky, but flows are also a lot higher for this scenario. In June and July monthly 2050's low scenario flows are lower than the baseline scenario. Low summer flows can also be seen with the

2050's high scenario, although these are slightly higher than those shown by the baseline and low scenario.

Figure 8.3.2 shows the comparison between TP concentrations for the baseline and 2050's high and low climate scenarios. The graph shows that the high scenario results in the highest TP concentrations. All scenarios follow the same annual pattern, with lower TP concentrations occurring in the summer months. It is during the summer months that there is the smallest difference between the three scenarios. All of the scenarios breach the lower threshold limit (0.1 mg l^{-1}) for ecological failure as suggested by Severa-Martinez (2005) and the EA water quality target (0.1 mg l^{-1}) (EA, 2006), but none of the scenarios reach the upper threshold limit of 0.25 mg l^{-1}.

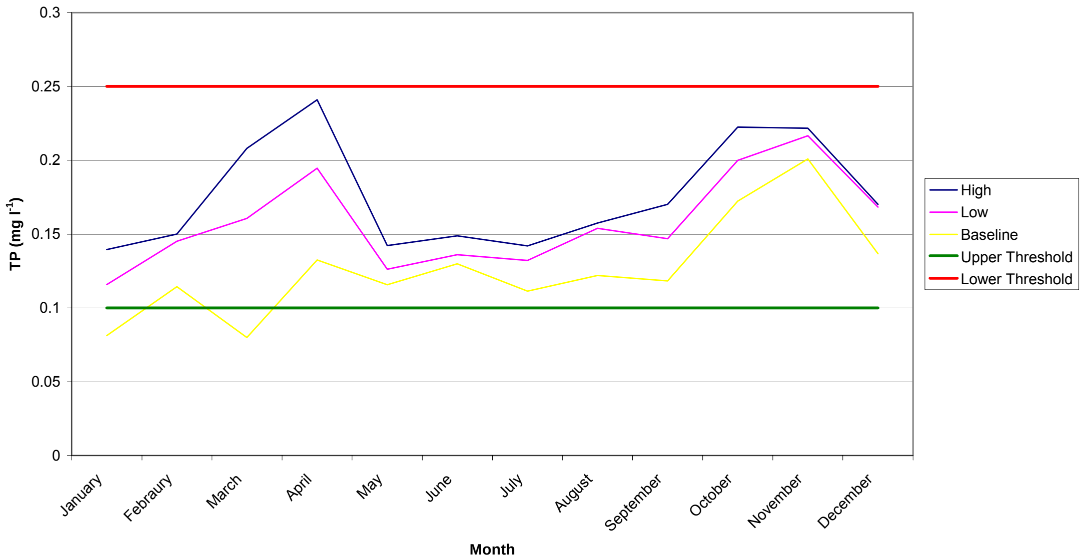


Figure 8.3.2: Comparison between average monthly TP concentrations (Bure and Ant) for baseline and 2050's high and low scenarios

Figure 8.3.3 shows monthly nitrate concentrations for all three scenarios. The annual nitrate pattern follows the same pattern as flow and TP concentrations with high winter values and low summer values. Once again the highest nitrate concentrations can be seen with the 2050's high scenario. Both the baseline and 2050's low scenario have comparable summer concentrations. The high scenario nitrate concentrations do not follow the exactly same annual pattern as the baseline and low scenario. Nitrate concentrations only reach the summer low values seen in the other two scenarios in

August. In comparison the summer low values plateau out between June and September. None of the scenarios breach the EA water quality target of 20 mg l^{-1} (EA, 2006).

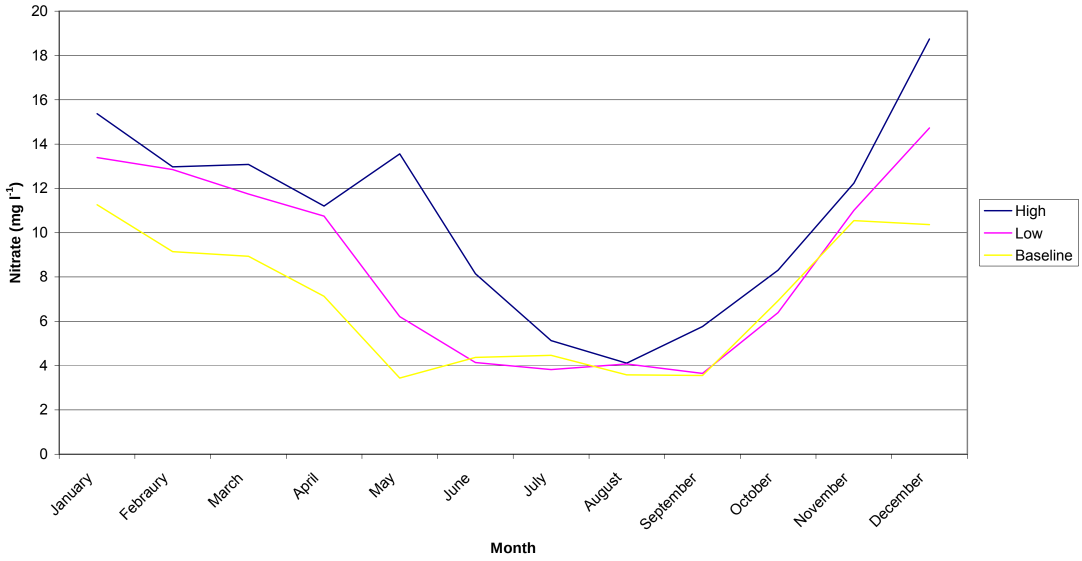


Figure 8.3.3: Comparison between SWAT nitrate concentrations (Bure and Ant) for baseline and 2050's high and low climate scenario

The following 8 Figures (8.3.4 – 8.3.11) show the spatial results of SWAT model runs with current land use and future climate scenarios for both the Bure and Ant and Thurne SWAT models in terms of loadings to each of the systems.

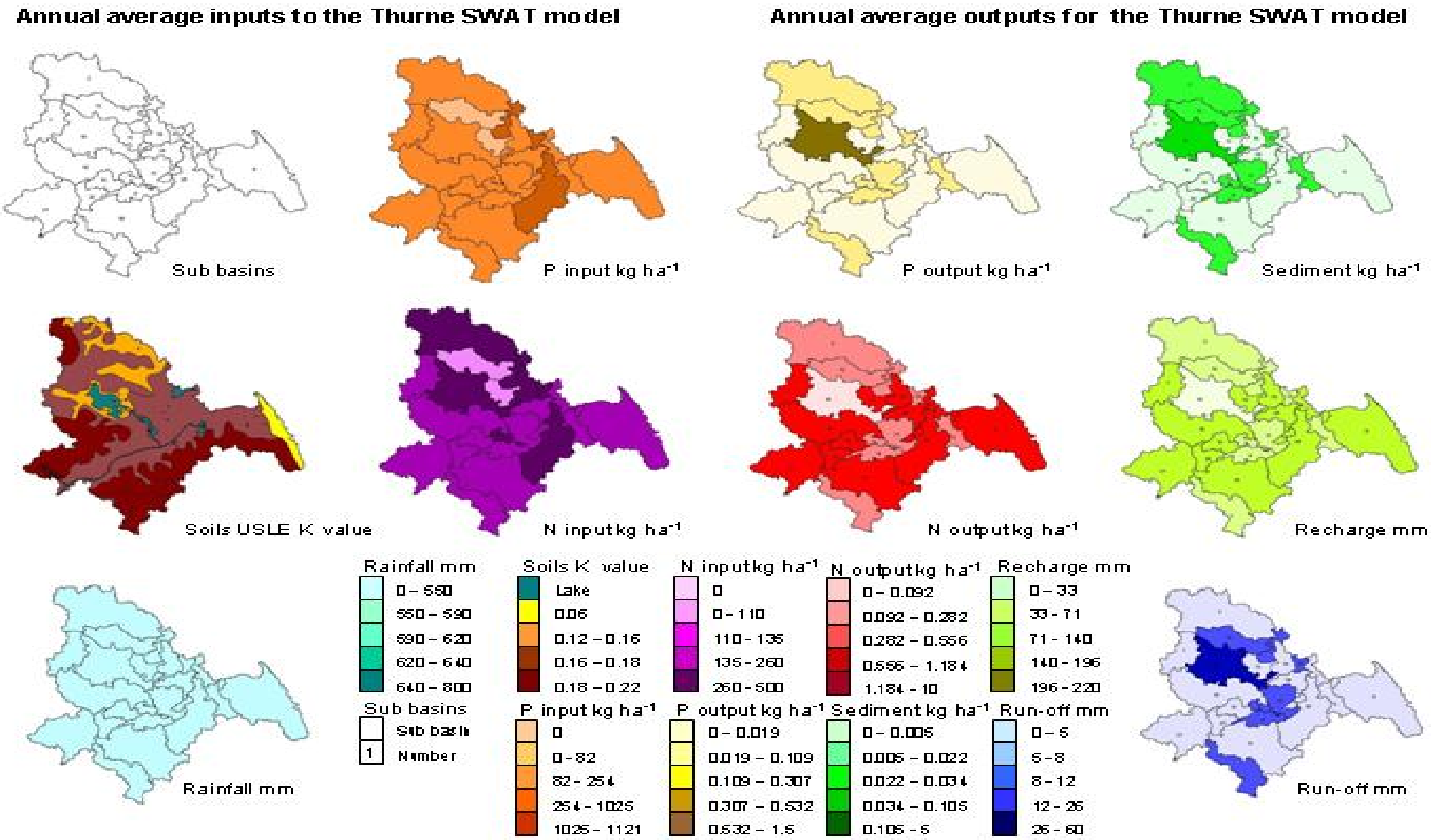


Figure 8.3.4: SWAT results for the Thurne model with current land use and current climate

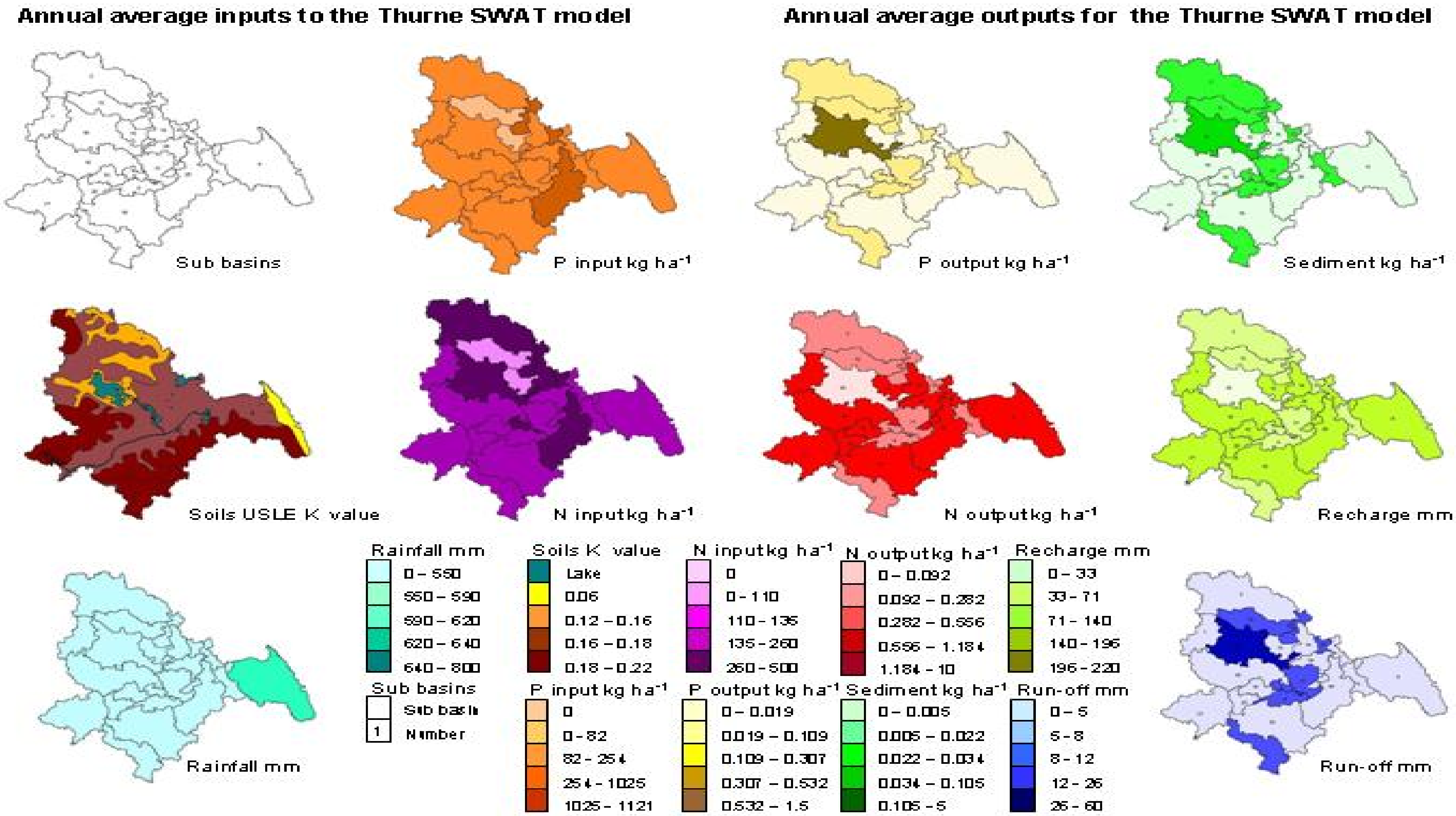


Figure 8.3.5: SWAT results for the Thurne model with current land use and baseline climate scenario

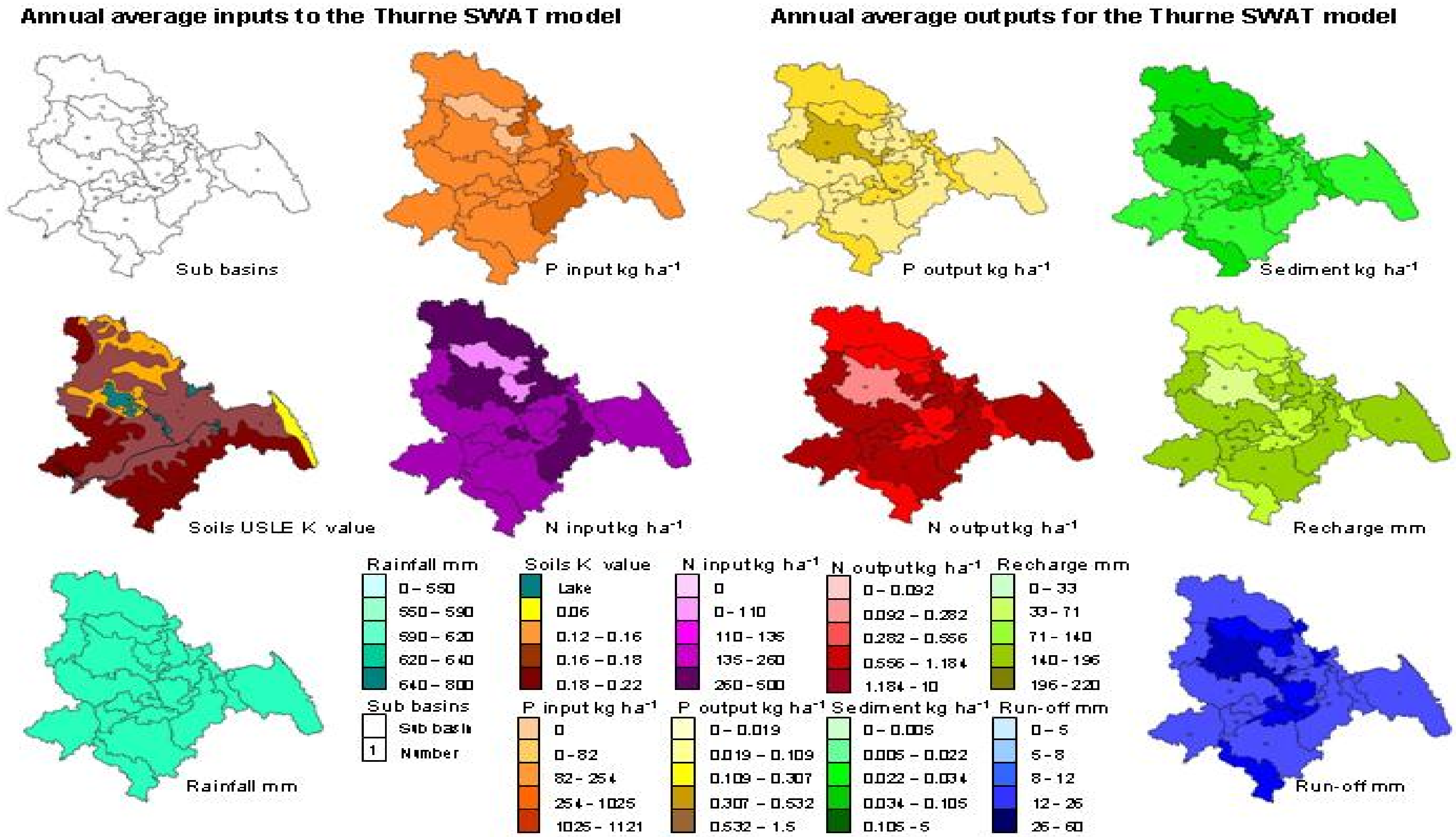


Figure 8.3.6: SWAT results for the Thurne model with current land use and low climate scenario

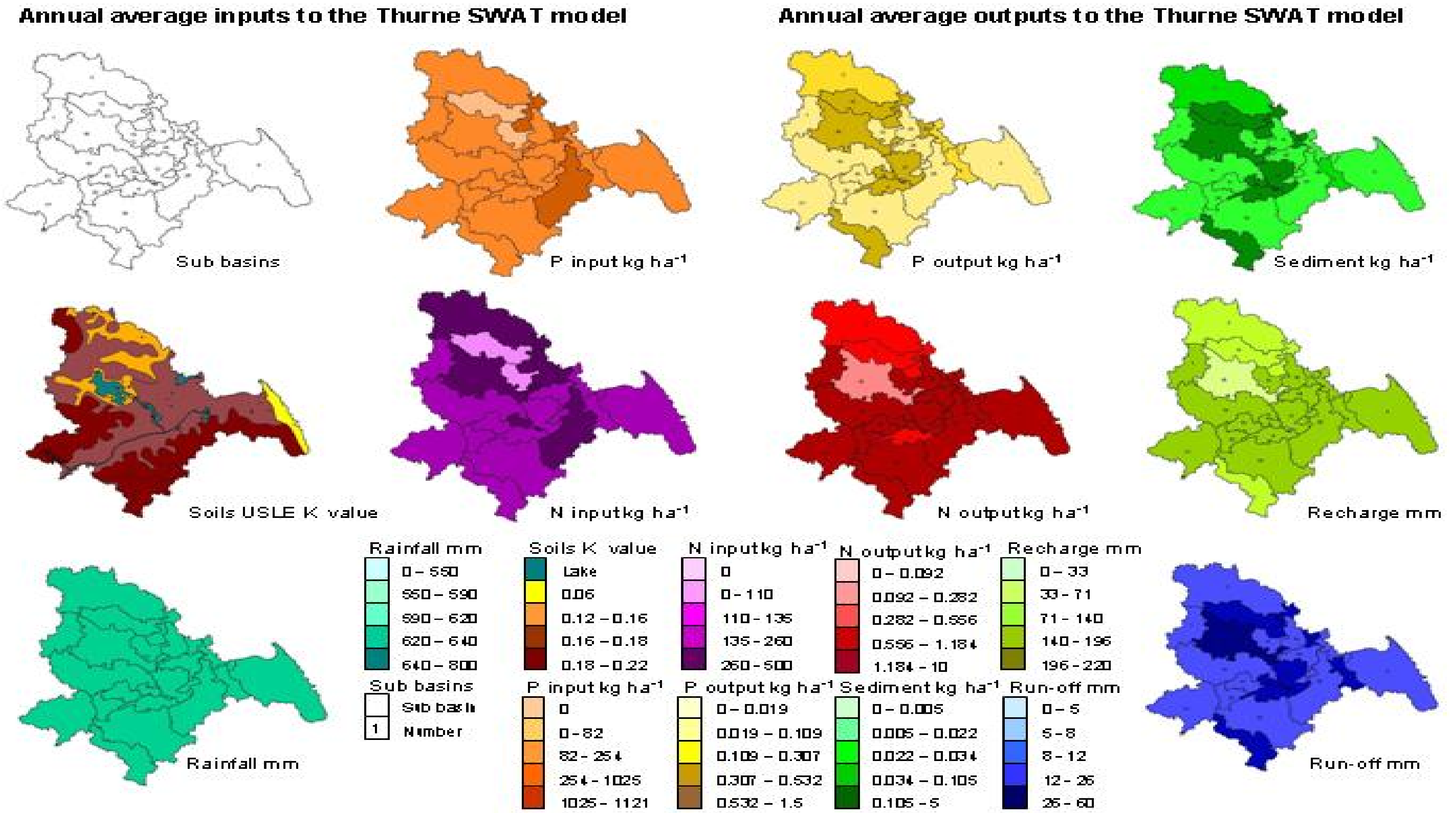


Figure 8.3.7: SWAT results for the Thurne model with current land use and high climate scenario

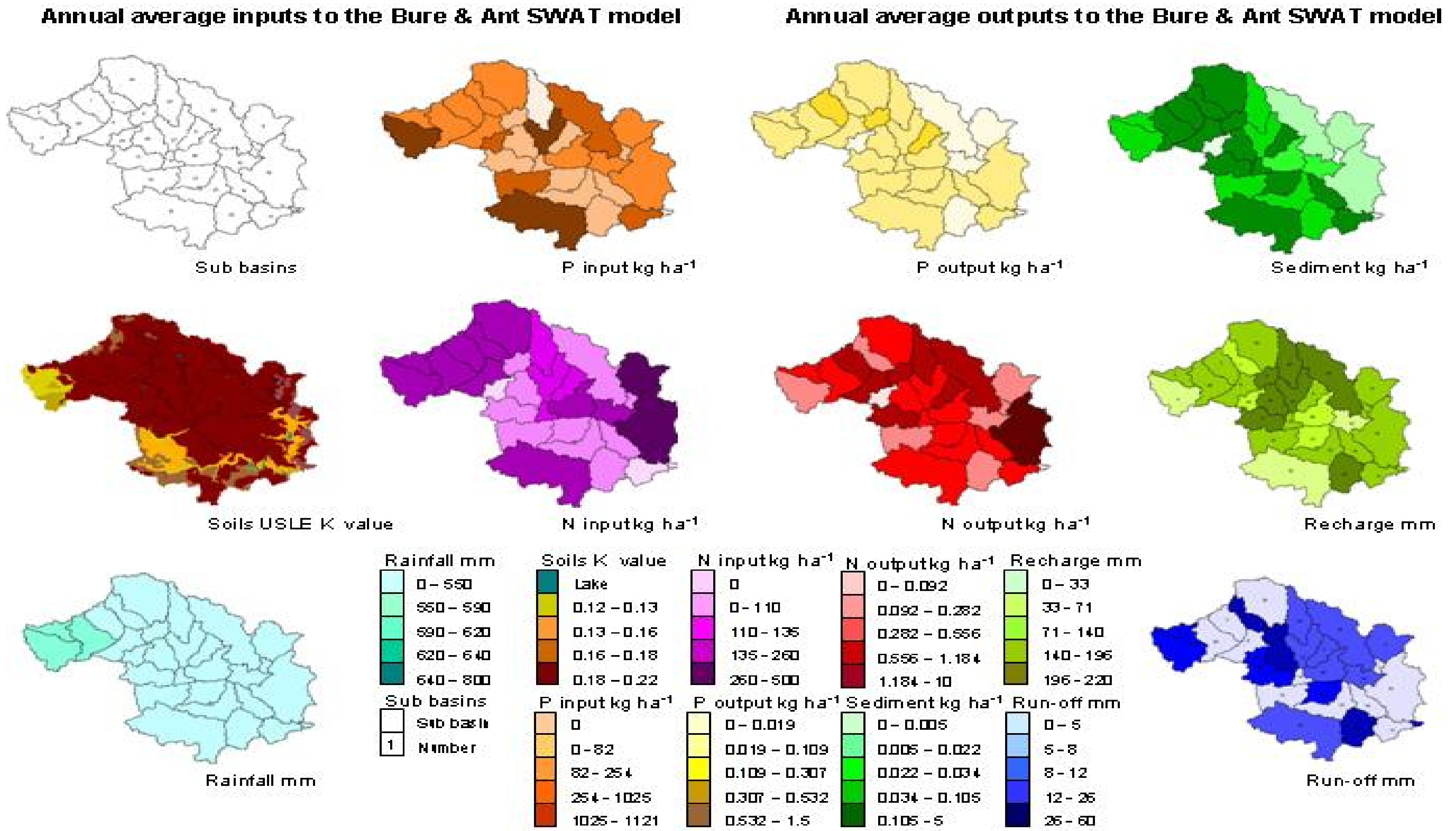


Figure 8.3.8: SWAT results for the Bure & Ant model with current land use and current climate

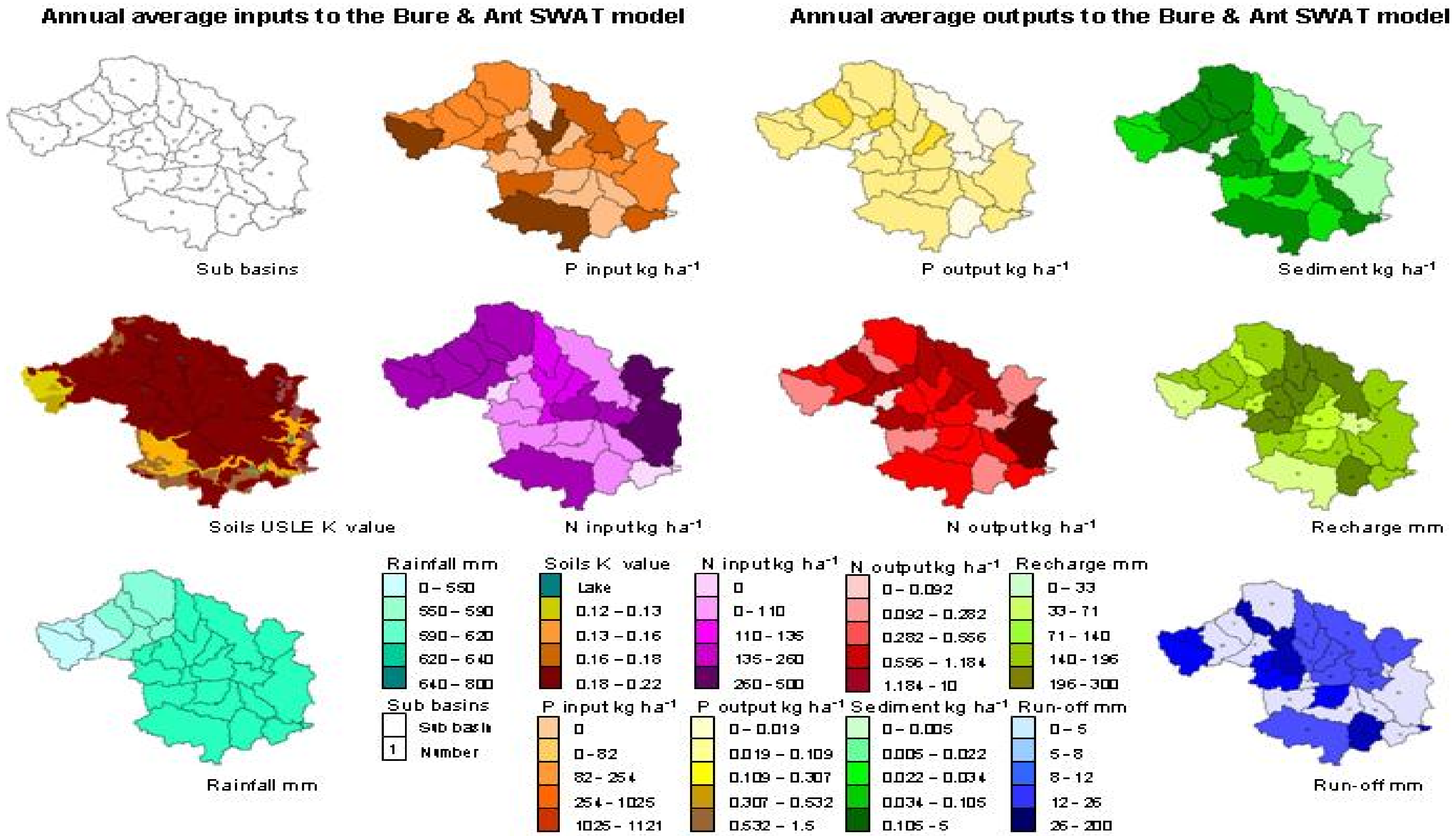


Figure 8.3.9: SWAT results for the Bure & Ant model with current land use and baseline climate scenario

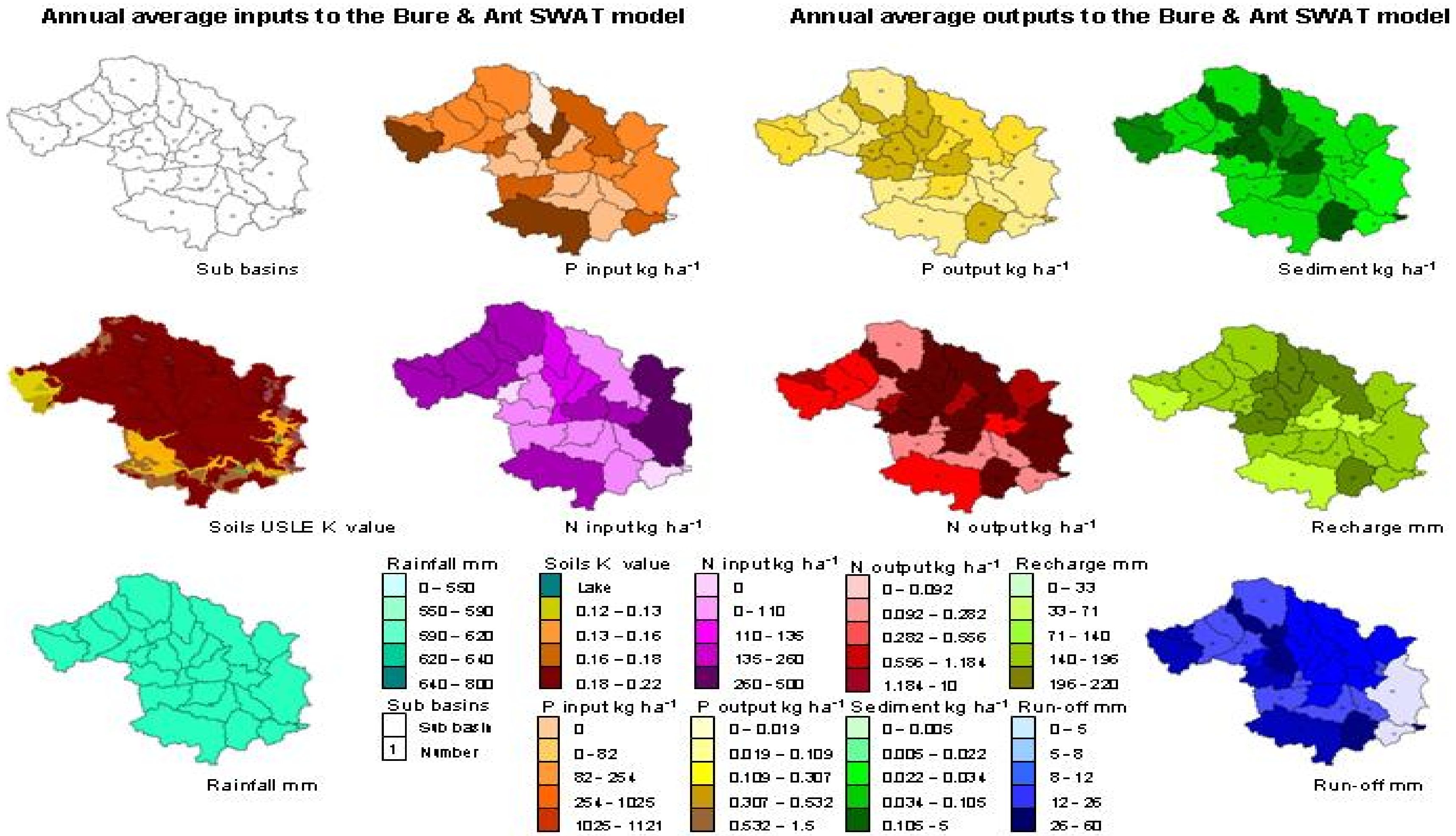


Figure 8.3.10: SWAT results for the Bure & Ant model with current land use and low climate scenario

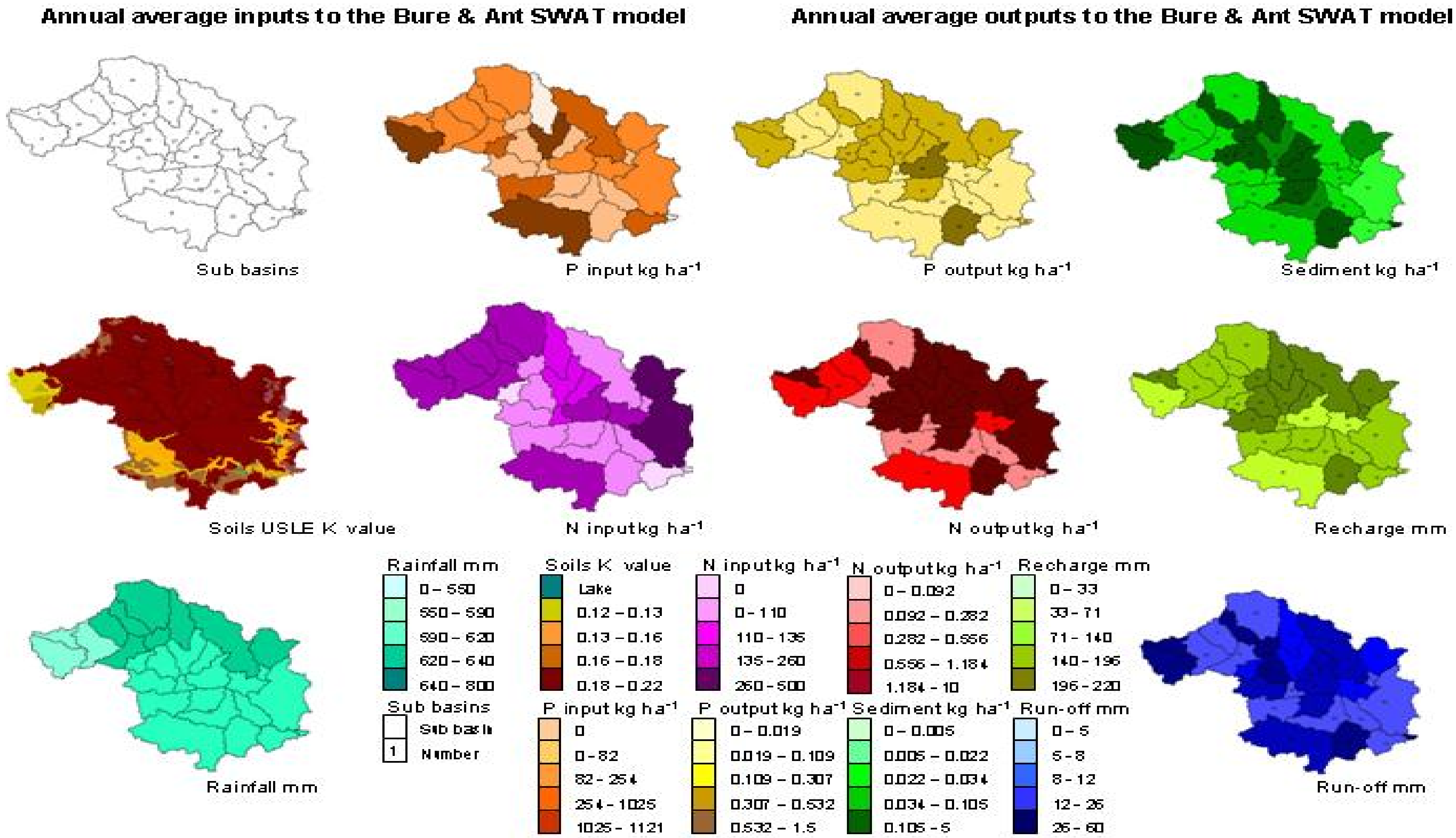


Figure 8.3.11: SWAT results for the Bure & Ant model with current land use and high climate scenario

By considering the spatial distribution of input and output loadings to the two SWAT models a number of patterns can be seen. Under observed and baseline rainfall there is lower P output for both the catchment systems when compared to the 2050's high and low climate scenarios. This occurs even though all the climate scenarios are receiving the same P input for both the modelled systems. N outputs behave in a similar way, with more sub basins showing the higher N outputs.

For both the future climate scenarios the increased rainfall results in increased re-charge values. Where infiltration values increase there is an increase in N loadings to the catchment systems, suggesting the main source of N yields in the study area is through groundwater. Where there are lower infiltration values there is higher run-off values resulting in higher P and sediment yields within the study area.

Recharge with SWAT is governed by groundwater and soil parameters. Groundwater parameters are the recharge coefficient (GW_REVAP) and the threshold water level in the shallow aquifer for percolation to occur (REVAPMN) both of these parameters are used within calibration and can therefore not be used to explain causes of high recharge values within the two catchment systems. The main soil parameter, which can affect recharge, is the saturated hydraulic conductivity (K_{sat}) which has been taken from the LandIS database. Within Figure 8.3.7 in can be seen that sub-basin 9 has a high recharge value (140 – 196 mm) and sub-basin 2 a lower value (71 – 140 mm). The soil in sub basin 9 is predominantly the Wick association which has a K_{sat} value of 82 mm hr^{-1} where as the Gresham association in sub basin 2 only has a K_{sat} value of 55 mm hr^{-1}, therefore soil water moves more slowly through the soil affecting the recharge of the sub basin.

The spatial distribution for each output parameter does vary depending on the climate input, as this is the only factor, which is being changed. Similar patterns can be seen throughout the results. Higher recharge areas occur in the south of the Thurne model through all of the scenarios, and within the Bure and Ant model higher recharge values occur within the centre of the catchment. However a spatial relationship between rainfall and run-off cannot be clearly seen. This may be caused by the lack of spatial relationship between the MONARCH grid cells used in the baseline and future scenarios, even though data were scaled based on one gauge in the study area. The results of the scaling procedure gave ratios, which were very close for all the MONARCH grid cells used, and consequently a fairly uniform rainfall throughout the two catchment systems.

Within the Bure and Ant system it can be seen that where high sediment outputs occur there are also high P yields as a result of high run-off. This can especially be seen in Figure 8.3.11, where sub basins 21 and 22 both have high P output, even though some of the sub basins have low P input. These sub basins all also have high sediment losses and run off values. This pattern can also be seen clearly within the Thurne system. Areas of high run-off and sediment yields can be attributed to the topography of the sub-basin, for example sub-basins 21 and 22 (Fig 8.3.11) have longer slope length (72 m and 111 m respectively) and higher average slope values (0.43 m m^{-1} and 0.55 m m^{-1} respectively) over the whole sub-basin. The slope length and average slope parameters in SWAT are topographical factors derived from the DEM, which affect the ratio of soil loss per sub-basin. In comparison sub basin 16 (Fig 8.3.11) where the slope length is 42 m and the average slope is 0.02 m m^{-1}, shows less run-off and sediment output from the sub-basin.

Sediment erosion is also linked with the run-off curve number within SWAT. The curve number is a function of the soil's permeability, land use and antecedent soil water conditions. Within sub basins 21 and 22 (Fig 8.3.11) the run-off curve number is 76 as the soil has a moderate infiltration rate and is moderately well drained, small grained crops (winter wheat and barley) are being grown in the sub-basin in a straight row with good cover. In comparison in sub basin 16 (Fig 8.3.11) the run-off curve number is 67 as the soil has low run-off potential with a high infiltration rate, peas are being grown in the sub basin in a straight row with good cover, resulting in lower sediment yields from the sub basin.

The results from SWAT show that the 2050 high climate scenario has the greatest impact on both the systems, with increased N, P and sediment output along with high recharge and high run-off. This can be seen more clearly in Tables 8.3.1 and 8.3.2.

Table 8.3.1: Average SWAT outputs for current and future climate scenarios under current land use model for the whole of the Bure and Ant system

Climate Scenario	Rainfall (mm)	Run-off (mm)	Sediment (kg ha^{-1})	N (kg ha^{-1})	P (kg ha^{-1})	Recharge (mm)	Flow (m^3 s^{-1})
Current	519	12.37	0.05	0.59	0.10	123.75	1.69
Baseline	520	15.65	0.09	0.62	0.13	126.45	1.71
Low	586	19.54	0.09	1.01	0.15	142.88	1.88
High	616	23.48	0.22	1.11	0.20	162.34	2.00

Table 8.3.2: Average SWAT outputs for current and future climate scenarios under current land use model for the whole of the Thurne system

Climate Scenario	Rainfall (mm)	Run-off (mm)	Sediment (kg ha^{-1})	N (kg ha^{-1})	P (kg ha^{-1})	Recharge (mm)	Flow (m^3 s^{-1})
Current	502	7.40	0.06	1.34	0.83	138.26	0.96
Baseline	504	7.41	0.07	1.36	0.86	139.59	0.96
Low	613	13.11	0.10	1.52	1.02	153.28	1.06
High	627	17.06	0.22	1.61	1.18	163.05	1.16

The above tables show that overall the Thurne system has greater nutrient output values then the Bure and Ant system. This difference is greatest for P loadings with the low and high scenarios being greater by 15% and 17% respectively. The higher N values found in the Thurne catchment can be attributed to the high recharge values occurring in the Thurne system.

By looking at nutrient loadings within the catchment it can be seen that in order to reduce nutrient loads then run-off and sediment erosion need to be reduced. Undertaking soil conservation practices within the catchment systems such as the use of cover crops and conservation tillage may reduce run-off. The uptake of N by plants also needs to be increased to reduce the amount of N leaching through the soil. By using conservation tillage it maybe possible to keep nitrogen fertilisers in the upper layers of the soil where they will be available for plant uptake. The use of cover crops will reduce soil erosion over the winter months and increase N uptake during the winter months. It will also reduce the occurrence of nutrient ‘flushing’ in the spring months by the first big storm after fertiliser application. Erosion control practices will be discussed further in section 8.4.

8.3.1 Results of Future Land Use and Climate Model Runs

Figures 8.3.12 – 8.3.15 show the spatial results of SWAT model runs with future land use and future climate scenarios for both the Bure and Ant and Thurne SWAT models in terms of loadings to each of the system. Within the two catchment systems it can be seen that the high RE scenario results in high run-off, sediment, and nutrient yields. This is a result of the high rainfall scenario and the change in land use to a system, which is dominated by spring crops. This combination results in higher run-off curve values (from 0.67 in sub basin 16; Figure 8.3.11, to 86 in sub basin 16 Figure 8.3.15) as the soil is left bare during the winter months.

Recharge has also increased with the selected land management scenarios, although the increase is not so great when compared to the affect of the climate scenarios. Between the base line scenario and the high climate scenario recharge increased by 36 mm, in comparison between the high climate scenario and the high climate coupled with the RE scenario this increase is only 6 mm (Table 8.3.3). This suggests that recharge is affected by climate change more than a change in land use.

As with the climate change scenarios where high run-off occurs this results in higher sediment and P loads. In terms of P input to the catchment this has reduced compared to SWAT current land use as there is a decreased amount of winter cereals in the two catchment systems, both winter wheat and winter barley have an application of 100 kg ha^{-1} during the autumn before the crops are planted compared to only 60 kg ha^{-1} for sugar beet which has increased in area for the two future land use scenarios.

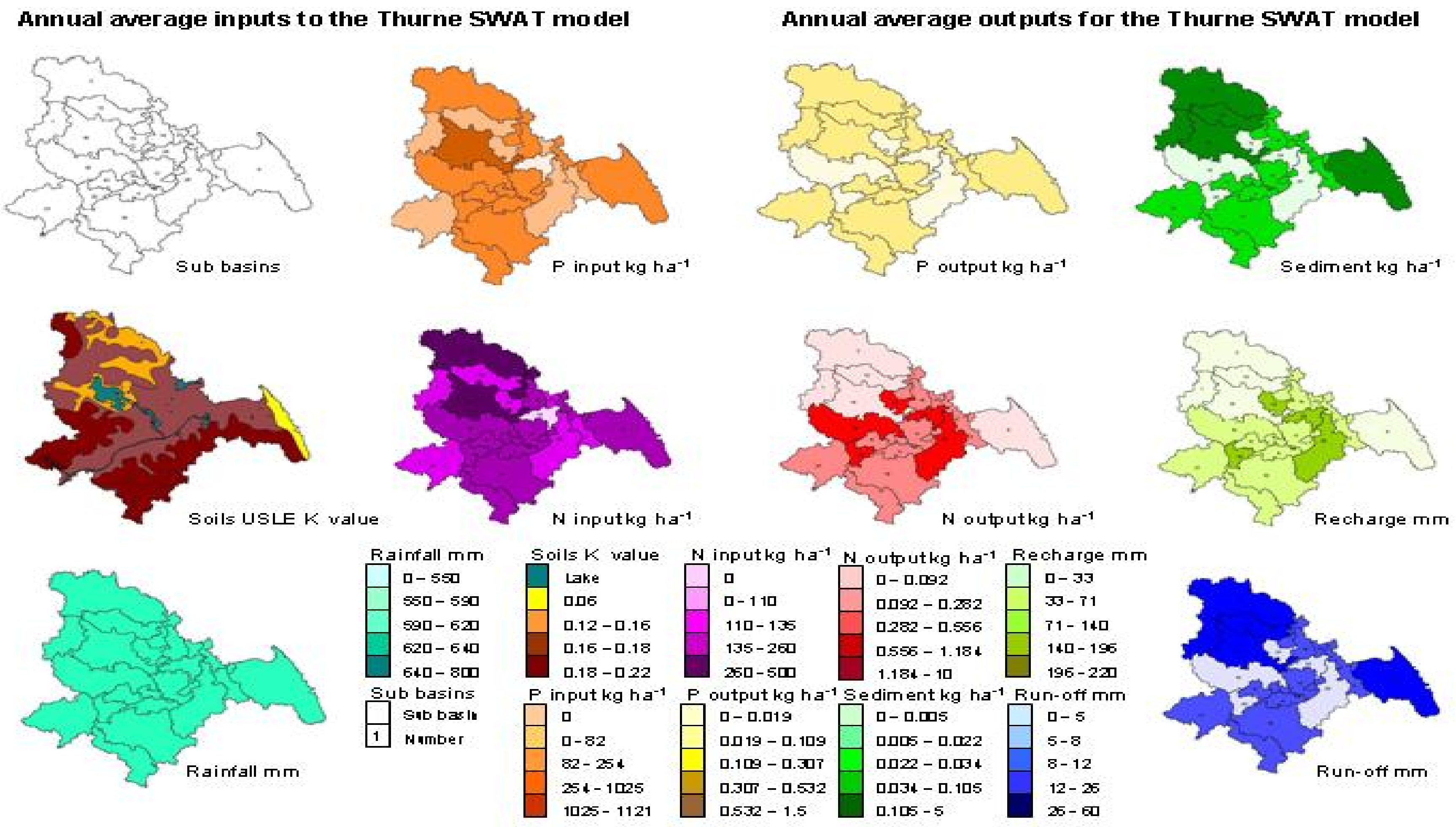


Figure 8.3.12: SWAT results for the Thurne model with GS land use and low climate scenario

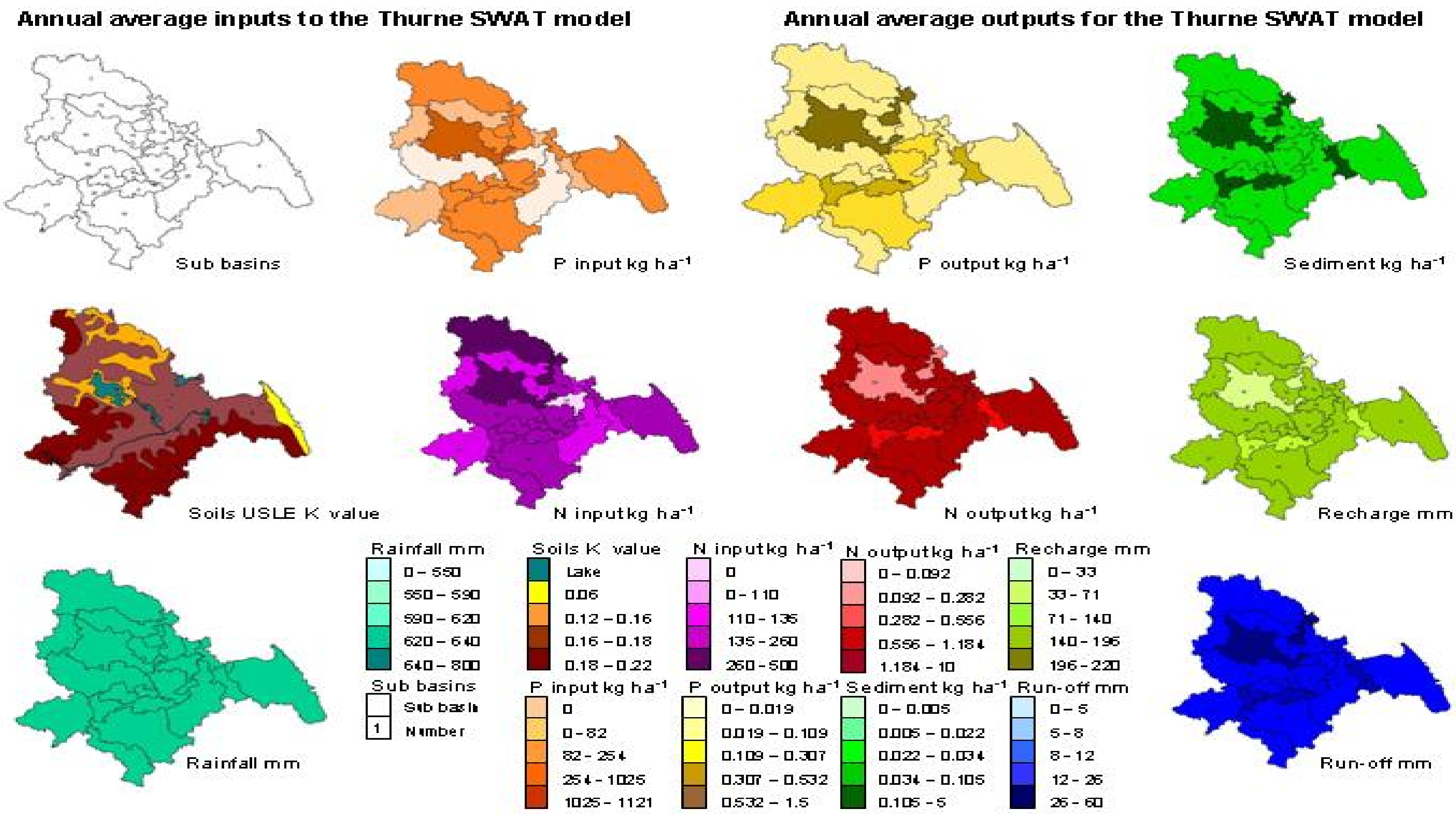


Figure 8.3.13: SWAT results for the Thurne model with RE land use and high climate scenario

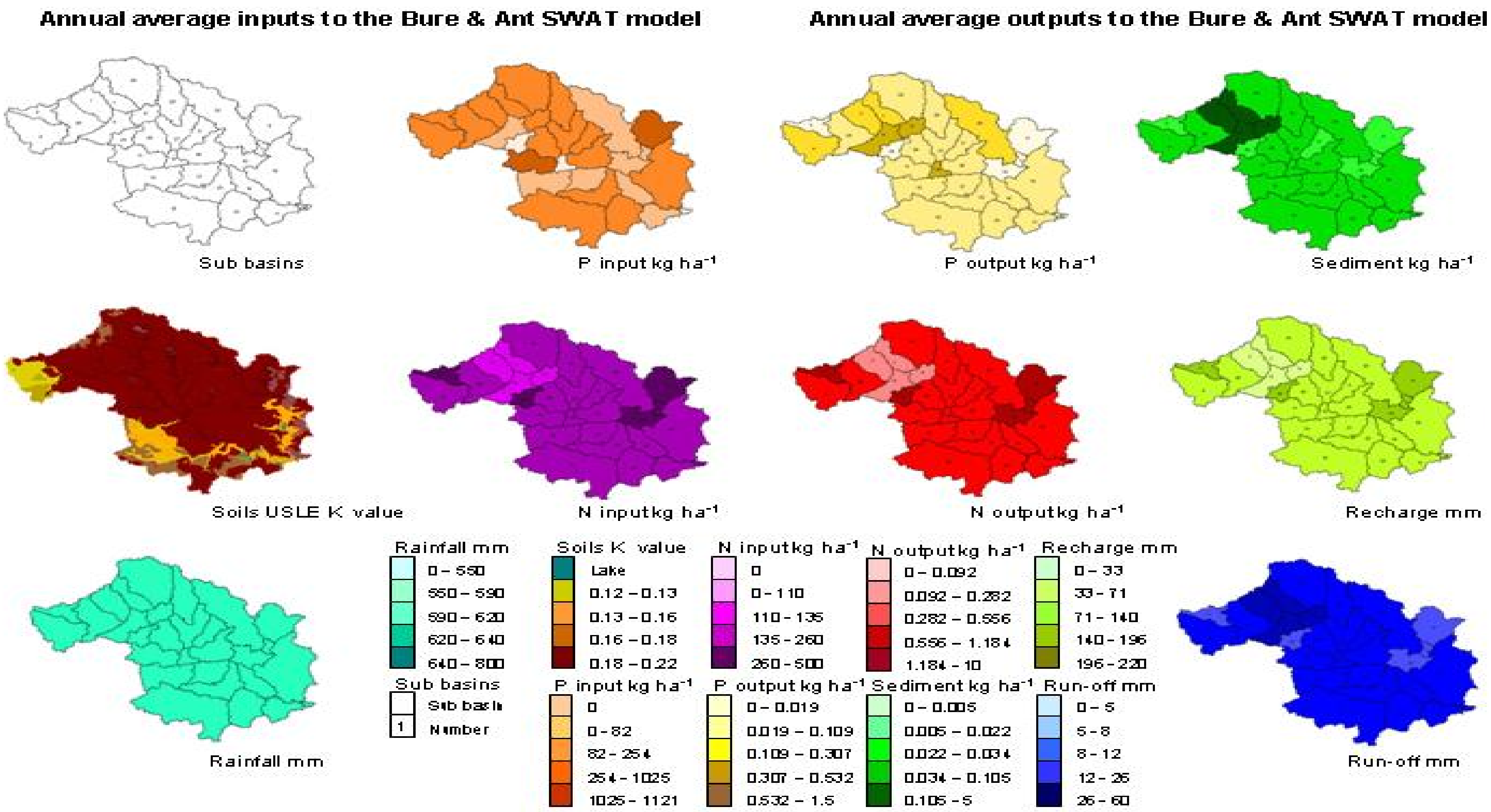


Figure 8.3.14: SWAT results for the Bure and Ant model with GS land use and low climate scenario

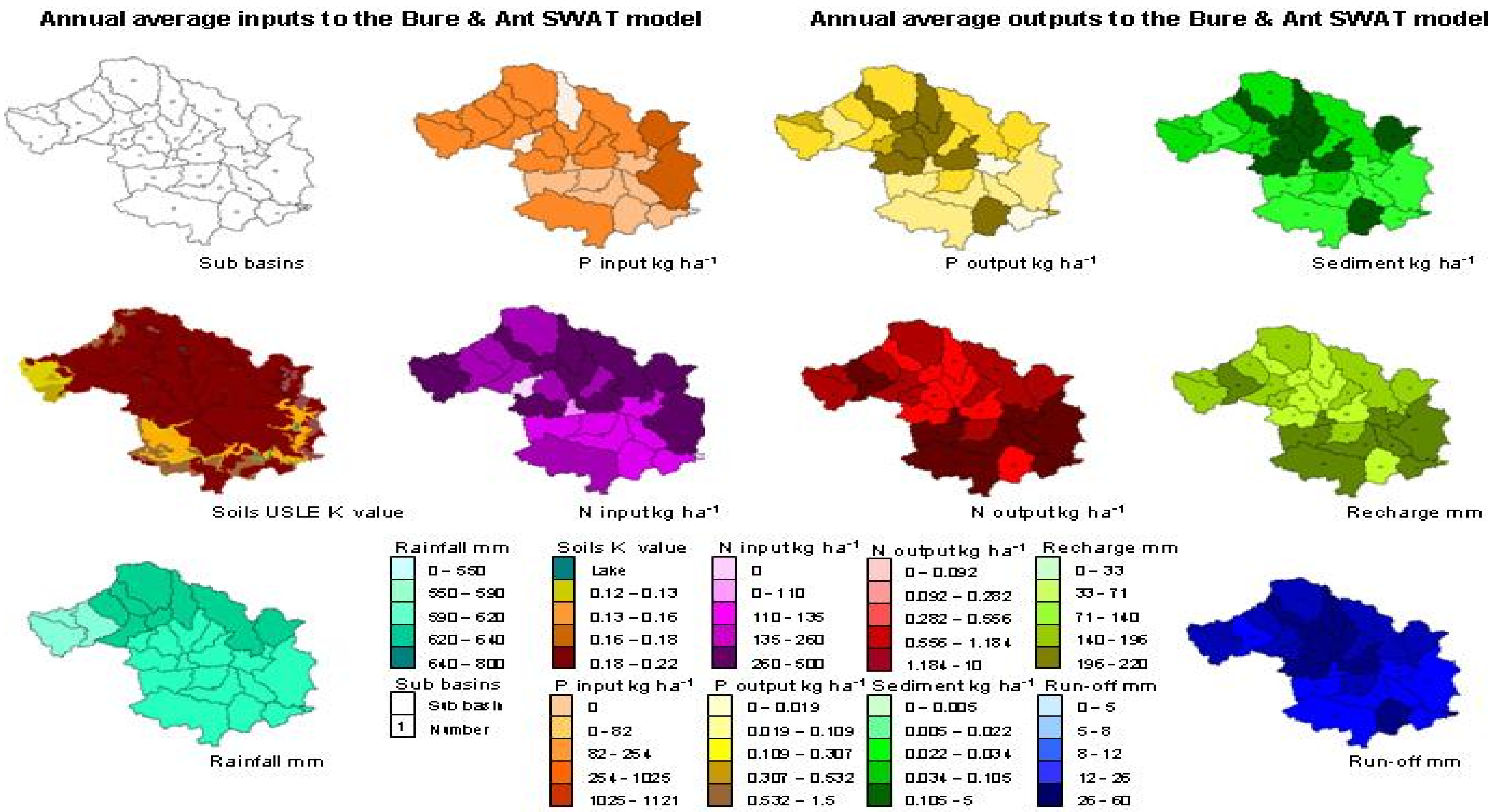


Figure 8.3.15: SWAT results for the Bure and Ant model with RE land use and high climate scenario

Table 8.3.3 shows that the GS land scenario actually improves the situation of the systems when coupled with the low climate scenario within both catchment systems. This may be due to a number of factors, a greater proportion of the two catchments are forested within this scenario. The Thurne catchment has a difference of 458 ha in forested area between the GS and RE scenarios. The GS scenario also results in a greater variety of crops being grown in the two systems compared to the baseline and RE scenario. This greater variety means that there is a better cover of crops resulting in lower run-off and sediment yields and improved uptake of nutrients.

Table 8.3.3: Average SWAT outputs for current and future climate scenarios under current and future land us for the whole of the Bure and Ant system

Climate Scenario	Rainfall (mm)	Run-off (mm)	Sediment (kg ha^{-1})	N (kg ha^{-1})	P (kg ha^{-1})	Recharge (mm)	Flow (m^3 s-^{1})
Current	519	12.37	0.05	0.59	0.10	123.75	1.69
Baseline	520	15.65	0.09	0.62	0.13	126.45	1.71
Low	586	19.54	0.09	1.01	0.15	142.88	1.88
High	616	23.48	0.22	1.11	0.20	162.34	2.00
Low GS	586	15.53	0.09	0.55	0.12	134.88	1.73
High RE	616	26.79	0.25	1.15	0.23	168.34	2.01

Table 8.3.4: Average SWAT outputs for current and future climate scenarios under current and future land us for the whole of the Thurne system

Climate Scenario	Rainfall (mm)	Run-off (mm)	Sediment (kg ha^{-1})	N (kg ha^{-1})	P (kg ha^{-1})	Recharge (mm)	Flow (m^3 s-^{1})
Current	502	7.40	0.06	1.34	0.83	138.26	0.96
Baseline	504	7.41	0.07	1.36	0.86	139.59	0.96
Low	613	13.11	0.10	1.52	1.02	153.28	1.06
High	627	17.06	0.22	1.61	1.18	163.05	1.16
Low GS	613	12.08	0.04	1.28	0.82	153.06	0.98
High RE	627	19.30	0.25	1.78	1.21	169.11	1.17

8.4 Management Solutions

From the climate change and socio-economic scenarios that have been simulated in SWAT, key source areas of nutrient loadings can be identified. Management scenarios can be applied to these source areas either at the river basin, or sub-basin scale, which will help to reduce nutrient loadings.

Management scenarios are based on different agricultural practices, which can be used to reduce nutrient loads to water. There are five main ways to manage nutrients, especially phosphorus.

1. Manipulation of poultry diets to reduce the amount of nutrients excreted in manure
2. Physical or chemical treatment of manure to separate some of the nutrients from the manure
3. Application of manure based on crop nutrient requirements, using methods that reduce the risk of run-off to surface waters
4. Effective soil erosion control practices on application sites including no-till agriculture, contour tillage, leaving crop residues on the soil surface after harvest and growing winter cover crops
5. Use of vegetative buffer strips along stream and river banks to slow down run-off, capture sediments and increase infiltration and phosphorus uptake rates.

However, only methods 3, 4 and 5 can be applied to SWAT management files and method 3 is already being incorporated in the SWAT models through the use of recommended fertiliser application rates Therefore, management scenarios are based on methods 4 and 5.

There are a number of different methods of reducing soil erosion as mentioned above. Cover crops are grown to protect and improve the soil, not to harvest. Cover crops have the potential to improve soil tilth, control erosion and weeds and maintain water balance within the soil, which decreases leaching of nutrients. Cover crops retain and recycle plant nutrients (especially nitrogen) between crops, provide habitat for beneficial microorganisms and increase plant diversity. There are many ways to use cover crops in a production cycle.

- As a main crop during the primary growing season. Used as a rotational crop, the cover will exclude production of a cash crop.

- As a companion crop, or living mulch, the cover is planted between the rows of cash crop.
- As a 'catch' crop for nutrients, planted after harvest of the main crop or between the rows of the cash crop to reduce leaching of nutrients
- As an off-season crop grown to protect the soil, usually during the winter when there is no main crop. This is the most common practice

Vegetative buffer strips are vegetated areas along rivers and other sensitive areas such as wells where fertilisers and manures are not normally applied. The purpose of these strips is to form a physical barrier between the field and the surface water. Any run-off coming from the field will be slowed down and intercepted by the vegetation. This will not only reduce the speed of movement of the run-off, but also capture some of the sediments and large organic particles in the run-off. It will also promote infiltration and increase nutrient uptake. In addition to nutrient removal, buffer strips can provide secondary benefits, such as river stabilisation or refuge for wildlife species. Table 8.4.1 shows the benefits of the erosion and run-off practices and suitability for use in SWAT (Devlin *et al.,* 2002 and Mishra *et al.,* 2003).

Table 8.4.1: Erosion control practices adapted from Devlin *et al.* (2002) and Mishra *et al.* (2003)

Practice	Description	Benefit	SWAT Suitability
Conservation tillage	Cropping system that maintains at least 30% of the soil surface covered with residues after planting	Helps reduce erosion	Can be done through the use of SCS curve numbers. This is a function of the soil's permeability, land use and antecedent soil water conditions
Contour farming	Planting crops in rows that follow the contours of the land, perpendicular to the slope of the land	Reduces sheet and rill erosion	X River basin is very flat
Gradient terraces	A terrace designed to divert run-off to a suitable outlet, such as a grass waterway	Reduces speed of run-off and hence amount of soil that is eroded from the field	X
Level terraces	A terrace designed to store water until it can be passed through an underground outlet or seep into the soil	Stops water movement and allows eroded soil particles to settle out	X
Grass waterways	Sodded channel that provides an outlet for run-off	Reduces potential of gully erosion	X
Contour strip cropping	Alternating strips of close growing erosion resistant crops and erosion susceptible row crops, planted on the contour	Straps eroded sediments	Alternative crops can be grown through the use of HRU's within a sub-basin. No control on where they are grown
Vegetative filter strips	Strips of permanent vegetation on the down hill perimeter of erosive crop field or between the field and water body	Catches and filters sediments from surface run-off	Alternative crops can be grown through the use of HRU's within a sub-basin. No control on where they are grown
Constructed wetlands	Artificial wetland created down hill from crop fields	Sediments are collected and soluble nutrients are assimilated by growing vegetation	Alternative crops can be grown through the use of HRU's within a sub-basin. No control on where they are grown
Sediment control basin	A short earth embankment constructed across the slope to form a sediment basin	Traps run-off water and allows sediments to settle out	X
Critical area planning	Planting permanent vegetative cover on highly erodible lands that cannot be stabilised by conservation practices	Takes erodible land out of production	Can either plant permanent vegetation on a sub-basin or HRU scale
No tillage	Direct seeding of the crop into previous residue without tillage	Greatly reduces soil erosion and increases infiltration rates on most soils	Can remove tillage practices from SWAT management files
Cover crops	Cover crops are grown to protect and improve the soil, not to harvest	Protect and improve soil	Can incorporate cover crops into crop rotation schemes

From the above table in can be seen that not all the erosion run-off practices can be modelled in SWAT. Although the use of vegetative buffer strips can potentially be modelled in SWAT along with constructed wetlands and contour stripping there is no control on where these management solutions may be placed in an HRU or sub basin. Effective soil erosion practices such as critical area planting, no tillage and cover crops can however be modelled in SWAT with ease, and allow the user to control the placement of these measures within SWAT. Therefore these three erosion practices have been modelled for each of the future scenarios. This will allow the assessment of effective soil erosion practices on the study area, to see whether or not they will reduce nutrient and sediment loading the study area. These practices have been applied to sub basins which fall into the highest two categories for nutrient outputs as seen in Figures 8.3.1 to 8.3.12. Grass was grown for the critical area planting management solution without any fertiliser application throughout the whole study area to give a 'best' case scenario.

8.4.1 Management Solution Results

Table 8.4.2 shows the results of the three management solutions for the Bure and Ant SWAT model under the different future scenarios for varying parameters. It can be seen in all cases that critical area planting (pasture) throughout the study area is the best management solution. In terms of total phosphorus concentrations, critical area planting results in 45% reduction in concentration for the baseline scenario, this increases to a 58% reduction for the High RE future scenario. Although a reduction in total phosphorus concentrations can be seen for all the future scenarios under critical area planting the difference between the four future scenarios is very small, only varying between 0.10 - 0.15 mg l^{-1}.

The reduction in total phosphorus concentrations can be attributed to reduced sediment loads to the two catchment systems as P is mainly transported attached to sediment either as organic or mineral phosphorus. The reduction in TP concentrations with the use of cover crops is small ranging from 7 mg l^{-1} for the baseline scenario compared to current management conditions and no change for the high RE scenario. This can be attributed to the lack of reduction to sediment concentrations under cover crop management. The cover crop management solution does not reduce sediment concentrations under any of the future scenarios, with values being very similar to those of the current management scenario. This maybe because grass was chosen for the cover crop and did not give enough ground cover during the period between crop harvest and new crop planting dates to reduce sediment erosion. If another crop such as rye was chosen this may decrease sediment erosion as the crop provides longer and better erosion control because of more winter growth and a fibrous root system. This will reduce erosion, improve soil structure and reduce surface crusting and increase the water-holding capacity of the soil.

A reduction in TON concentrations can also be seen, the largest reductions once again being seen under the baseline scenario and critical area planting, 24%. Percentage reductions are lower for the future scenarios, 23% low, 15% high, 17% low GS and 24% high RE. The high RE scenario is the worst scenario for all the management solutions. This is especially so for the sediment concentrations and run-off results under all the management solutions except for critical area planting.

If the land use in the study area was to stay under a predominantly agricultural regime then the combination of both cover crops and no tillage management practices gives the best results as they will decrease soil erosion through improved water permeability, bulk density and aggregate stability. Therefore run-off values decrease under their combination of management solutions, especially for the high RE scenarios. However run-off values can be greatly reduced under the critical area planting solution for all scenarios as the ground is covered for the whole year, with plants growing close together, therefore reducing run-off and sediment erosion and improving infiltration. Nutrient loads are also greatly reduced as no fertilisers or manures are applied.

Table 8.4.2: SWAT results for management practices in the Bure and Ant model

		Average TP (mg l^{-1}) Bure & Ant outlet			
Scenario	Current Management	Cover	Cover & No Till	No Till	Pasture
Base line	0.18	0.11	0.13	0.12	0.10
Low	0.20	0.17	0.19	0.19	0.14
High	0.27	0.27	0.23	0.23	0.15
Low GS	0.19	0.18	0.20	0.20	0.14
High Re	0.36	0.36	0.29	0.28	0.15
		Average TON (mg l^{-1}) Bure & Ant outlet			
Scenario	Current Management	Cover	Cover & No Till	No Till	Pasture
Base line	6.04	6.58	6.68	6.73	4.63
Low	7.86	7.74	7.08	7.55	6.06
High	7.68	7.62	7.66	7.89	6.56
Low GS	7.28	7.20	7.05	7.12	6.06
High Re	8.54	8.45	8.17	8.26	6.56
		Average Flow (m^3 s^{-1}) Bure & Ant outlet			
Scenario	Current Management	Cover	Cover & No Till	No Till	Pasture
Base line	1.71	1.65	1.67	1.62	1.57
Low	1.88	1.78	1.75	1.77	1.72
High	2.00	1.93	1.86	1.92	1.83
Low GS	1.73	1.70	1.65	1.60	1.56
High Re	2.01	1.99	1.96	1.98	1.93
		Average Run-off (mm) Bure & Ant outlet			
Scenario	Current Management	Cover	Cover & No Till	No Till	Pasture
Base line	15.65	14.52	8.02	14.33	1.20
Low	19.54	17.53	10.43	17.00	1.50
High	23.48	20.09	11.97	19.69	2.00
Low GS	15.53	14.10	7.06	13.96	1.20
High Re	26.79	22.14	12.13	20.13	4.00
		Average Sediment (mg l^{-1}) Bure & Ant outlet			
Scenario	Current Management	Cover	Cover & No Till	No Till	Pasture
Base line	19.04	19.05	18.71	18.73	16.04
Low	26.68	26.41	22.02	25.95	14.84
High	33.59	33.58	25.61	25.98	16.14
Low GS	23.41	23.02	22.74	22.89	14.84
High Re	53.74	53.65	30.15	30.07	16.14

Within the Thurne model the effect of the management solutions have been considered on Hickling Broad. In terms of TP there is very little difference between the management solutions in any of the future scenarios, critical area planting does however achieve the lowest TP concentrations in the Broad. All of the management solutions reduce TP concentrations to below the lower TP threshold of 0.1 mgl^{-1} (Severa-Martinez, 2005) and EA's river water quality target of 0.1 mgl^{-1} based on the General Quality Assessment Scheme (EA, 2004), which is not being achieved under the current modelled management regime. As with the other parameters, NO_3

concentrations in Hickling Broad are reduced the most under critical area planting, by as much as 50% in the majority of future scenario cases and therefore do not breach the EA's river water quality target of 20 mgl^{-1} based on the General Quality Assessment Scheme (EA, 2004). There is however no clear management solution if the land in the Thurne watershed remained in an agricultural regime for both TP and NO_3. This is particularly so for TP concentrations which stay the same for all the management solutions except for critical area planting.

Table 8.4.3: SWAT results for management practices in the Thurne model

		Average TP ($mg\ l^{-1}$) Hickling Broad				
Scenario	**Current Management**	**Cover**	**Cover & No Till**	**No Till**	**Pasture**	
Base line	0.10	0.03	0.03	0.03	0.02	
Low	0.11	0.05	0.05	0.05	0.04	
High	0.12	0.06	0.06	0.06	0.05	
Low GS	0.11	0.04	0.04	0.04	0.03	
High Re	0.13	0.07	0.07	0.07	0.06	
		Average NO_3 ($mg\ l^{-1}$) Hickling Broad				
Scenario	**Current Management**	**Cover**	**Cover & No Till**	**No Till**	**Pasture**	
Base line	0.85	0.80	0.49	0.54	0.54	
Low	0.98	0.51	0.76	0.77	0.49	
High	1.01	0.96	0.59	0.81	0.59	
Low GS	0.83	0.64	0.64	0.67	0.43	
High Re	1.32	0.96	0.79	0.83	0.60	

8.5 Implications for the Study Area

There are a number of implications for the study area in terms of future climates. When compared to the current modelled situation in the two systems all of the future scenarios have an adverse effect. Overall the increase in rainfall in the study area leads to an increase in ground water recharge, run-off and river flows. Where there is high recharge in the two systems there is increased N yields being transported to the reaches via groundwater. This is demonstrated in the Thurne system where there are higher recharge values in the south of the system and consequently high N outputs.

Results show that where there are higher USLE K values for the soil higher sediment yields occur. Higher sediment yields can be expected for the future scenarios. Spatial results show that higher P outputs are linked to areas of higher run-off and consequently higher sediment loading, no matter how low the P input to the system.

This is because the majority of P is transported from the two systems attached to sediment in either mineral or organic form. Therefore P fertilisers should only be applied to the systems, especially the Thurne, where there is a low run-off and sediment yield. The impact of phosphorus concentrations from the future scenarios on Hickling Broad can be seen in Figure 8.5.1.

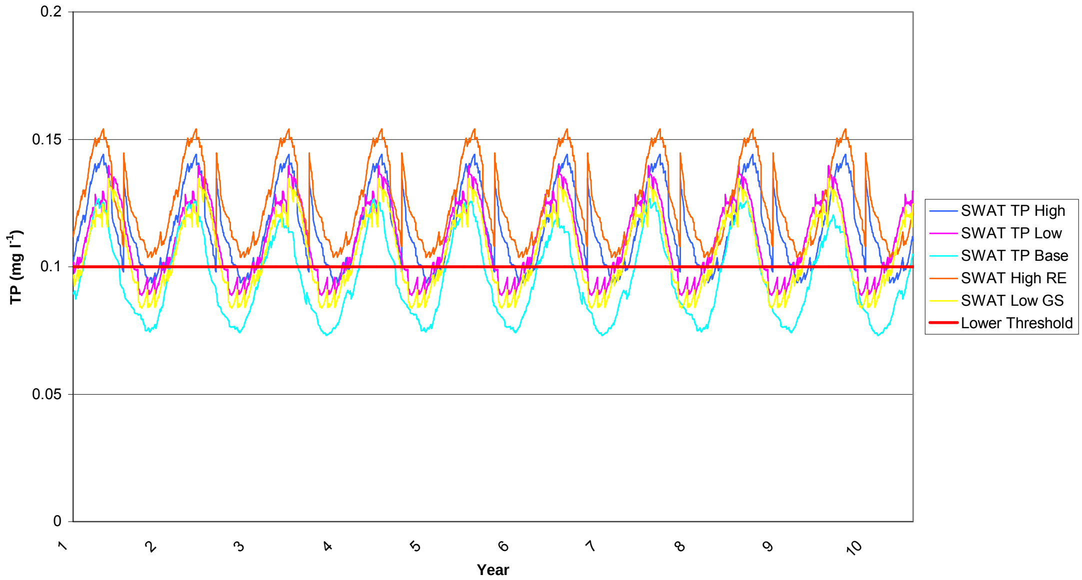


Figure 8.5.1: Total phosphorus in Hickling Broad for baseline and future scenarios

It can be seen for all the climate scenarios that total phosphorus concentrations breach the lower threshold limit for phosphorus of 0.1 mg l^{-1} (Severa-Martinez, 2005) and EA river water quality target 0.1 mg l^{-1} under the General Quality Assessment Scheme (EA, 2004) in the summer months, this even occurs in baseline conditions. Under the RE high climate scenario this value is breached throughout the year therefore increasing the likelihood of ecological failure within the Broad.

There is little difference between the major crops and the distribution of crops for the two socio-economic scenarios except for forested areas. The effect of the GS socio-economic scenario is to increase all the other break crops at the expense of cereals, with a system still very much in favour of wheat. The effect of the RE socio-economic scenario is to eliminate all break crops other than sugar beet and potatoes while barley and oats increase and wheat reduces. Therefore the effect of the two scenarios is very different with the GS scenario decreasing the nutrient loads to the two catchment systems below that of current conditions. However both scenarios also show a substantial increase in potential demand for irrigation due to the increased area of sugar beet and potatoes; this is however assuming unlimited water supplies.

George (1992) suggest that flow rates in the Broadland rivers are, from the ecological point-of-view, already less than they should be during the summer months. Consequently there has been a rise in the concentration of pollutants carried by these rivers, and a reduction in the rate at which broads associated with them are flushed by 'new' water derived from the catchment. This has resulted in blue-green algae blooms within the Broads. Increased demand for water from irrigation needs coupled with low flows in the summer months for all scenarios (Fig 8.3.1) could exacerbate this problem further. However climate scenarios show increased nitrate and TP concentrations during the winter months where flow is not a problem. It is only with the 2050's high climate scenario that there is a significant increase in nutrient concentrations, especially nitrate in the summer months (Figs 8.3.2 and 8.3.3). If water demand for irrigation and other industries was to increase then this could have very damaging effects on the region's biodiversity and future well-being.

Results show that the RE socio economic scenario coupled with the high climate scenario has the greatest adverse affects on the study area with increased run-off, sediment yields and P and N loadings. However it is climate change that causes the greatest increase in all these parameters by changing the hydrological regime of the study area. A 34% increase in run-off, 60% increase in sediment loadings, 45% increase in N loadings and 35% increase in P loadings to the Bure and Ant catchment can be seen with the 2050 High climate scenario. Lower increases can be seen in the Thurne system with P and N loadings increasing by 28% and 16% respectively. However, the high increase in the Bure and Ant system will have an adverse affect on the Thurne system as Moss *et al.* (1989) suggests that as much as 40% of the nutrient loadings from the Bure and Ant watershed travels up into the Thurne system on the flood tide as far as Potter Heigham. It was not possible to model nutrients on the flood tide in SWAT as SWAT only represents one directional flow and therefore the impact of increased nutrient loads from the Bure and Ant system and their impact on the Thurne system could not be investigated.

The nutrient and sediment loading to the study area can be decreased in both the baseline and future scenarios through erosion control practices. Critical area planting is the most effective control measure investigated. The implication of this measure is however to take land out of agricultural use and revert it to grassland. This is particularly the case in the north of the Thurne system, the area upstream of Hickling Broad where run-off is high and groundwater recharge is low, resulting in higher sediment and consequently nutrient loads. Where areas of both systems are still used as agricultural land, erosion control measures are needed to reduce the effect of possible future climate and socio-economic scenarios. Measures should also be employed in the present situation if the risk of ecological failure to Hickling Broad and eutrophication problems in the two systems is to be managed.

In summary the implications of possible future climate and socio-economic scenarios on the study area are:

- Increase in rainfall and temperature through climate change increase the already high risk of ecological failure in Hickling Broad.

- Increase in rainfall and temperature through climate change increase eutrophication problems for both the rivers and Broads within the Bure and Ant system.

- Climate change will increase run-off, sediment erosion and river flows within the study area.

- Agricultural practices on land with high run-off and high sediment erosion will increase nutrient loads reaching the watercourses.

- An agricultural regime under the RE and high climate scenario is the worst-case scenario for the study area in terms of increased nutrient loads reaching the watercourses.

- The best-case scenario would be to revert all the agricultural land to grassland, especially within the Thurne system. Where land is still used for agricultural erosion, control measures should be put in place.

- Erosion control measures, particularly critical area planting in the north of the Thurne system, should also be employed in the present situation if the risk of ecological failure of Hickling Broad and eutrophication problems in the two systems is to be managed.

Chapter Nine Summary and Conclusions

9.0 General

On a global scale lakes play only a minor role in the hydrological cycle. They however have special importance owing to their dominance in the landscape, value as a resource (water supply, hydropower, irrigation and amenity), value as natural ecosystems and centres of biodiversity. The Broads are shallow, eutrophic lakes, probably the result of medieval peat workings (Lambert & Jennings, 1960), concentrated in the Ant, Bure, Thurne and Yare river valleys. Increased nutrient loading to this fragile ecosystem from both point and diffuse sources of nutrients has resulted in 41 out of the 44 Broads being in a less favourable ecological condition.

Sustainable management of the Broads is needed to achieve ‘good ecological potential’ under the Water Framework Directive (Directive 2000/60/EC). To achieve this in the Broads wiser management of the entire catchment as well as a cluster of local solutions is required (Moss, 2001) both under current climatic conditions and possible future climate and economic scenarios. The most economical and practical way of investigating potential solutions is through catchment scale modelling.

9.1 Soil Water Assessment Tool

The Soil Water Assessment Tool (SWAT) has been used in this research. A literature review showed that SWAT has become an important tool for river-basin scale studies and has been used extensively throughout the world as well as a number of studies within the UK. It was therefore judged that SWAT could be used to study the impact of future climate and land use scenarios and predict the impacts of these scenarios on flow and nutrient dynamics in the Broads.

From a user’s point of view the use of SWAT has had several advantages. The model is freely available and it has interfaces with GIS and Windows for easy extraction of input parameters and analysis of results. Despite it strengths SWAT also has some weak points. It requires an extensive set of data to run. Knowledge in the subject area and a basic level of training are required to parameterise this model and to handle the input and output files. These are however problems of all complex catchment models and are not unique to SWAT.

An overall weakness of the SWAT model is the use of equations that have parameters that are not directly measured. For example, the curve number equation, although used often to estimate run-off volumes, is highly uncertain due to the use of a parameter (i.e. the curve number) that had not been determined empirically for use in the UK, but is rather derived for use in the USA. In addition, the MUSLE, which is used for soil erosion simulation, is also uncertain because of the number of parameters in the equation that are set from qualitative information (e.g. soil type and ground cover).

9.1.1 Evaluation of the use of SWAT in the study area

The use of SWAT in this research has been discussed in depth in the previous chapters. In this section a summary of the main evaluation points to do with the use of SWAT in the study area will be given.

- SWAT results are comparable with other UK studies but not with larger catchment studies outside the UK. These results illustrate a possible limitation in the SWAT model when modelling small catchments.

It is typically the case that a model that performs acceptably well for hydrology may still have limitations in fully capturing sediment loads. This is because the accurate simulation of sediment processes on the land surface is difficult to capture due to the heterogeneous nature of a catchment and the relatively unrefined equations used to explain certain processes (e.g. MUSLE).

There were considerable differences between the results of calibration and validation of sediment within the SWAT model; this was attributed to varying factors.

- It was found that sediment yield varied significantly with changes in sub-basin size and location. These effects were attributed to increasing levels of aggregation on average sub-basin slope and on the proportion of the sub-basin delineated as cropland.

- There are a number of reported sediment sources within the study area which have not been incorporated into SWAT. This is due to lack of sediment data and the fact that SWAT can only model sediment sources and loadings from HRUs/sub basins and channel degradation/deposition.

Calibration and validation results and overall model performance were also greatly affected by shortcomings which arose in SWAT when inputting the data into the model.

- SWAT auto irrigation function does not work, therefore CropWat was used to simulate irrigation schedules and amounts for all crops. It was noted that both CropWat and SWAT do not reflect common UK irrigation practice resulting in the soil being at field capacity for varying times of the year. This problem therefore needs to be addressed when modelling irrigation within SWAT in future.
- The use of reservoirs to model the land drainage pumps were discussed in Chapter 7. The sub-basins, which feed into the drainage pumps, are not topographically defined. In SWAT the pump sub-basins are delineated topographically therefore they are different in size and shape to those of the actual pump sub-basins. This affects the amount of run-off each sub-basin will receive. SWAT was therefore only able to model reasonably well the flow pattern at one pump.
- The calibration of the pump sub-basins in terms of nutrients highlighted the problem with using the reservoir function to represent the pumps given the sparse data set available. SWAT could not be used to simulate results at a daily level with any degree of confidence. However SWAT was able to predict nutrient levels in Hickling Broad well.

Overall the use of SWAT within the study area indicated limitations in the predictive capability of the model, especially for sediment. There are many possible sources of these errors, which have been discussed throughout this thesis: lack of input data, over simplification of various factors in the model equations, non optimal calibration parameters and errors in observed output data. However, based on the current available data, the model demonstrated its utility as a tool to understand processes in the watershed and as a basis for effective management in the Bure, Ant and Upper Thurne watersheds.

9.1.2 Importance of input and calibration/validation data within SWAT

SWAT is a comprehensive model that requires a diversity of information in order to run; therefore great care has been taken in selecting data for use in the model set up. The results from input and calibration/validation data investigations are discussed in Chapters 5 and 6. This section gives a brief review of the main problems and results.

- Soil and crop type data are only available at a national level.
- The soil database in SWAT is made up of the dominant soil series for each association found in the study area, thus the database does not take into consideration the characteristics of the other soil series making up the association. By only modelling the soil associations, soil series are not incorporated into the model; therefore model results could be underestimating soil erosion.
- Sensitivity analysis of soil series and association data showed that SWAT responds in a realistic way to different soil and land cover combinations. It also showed that the difference between soils in terms of sediment yield is relatively low and therefore the use of the National soil map and soil associations has a limited influence on SWAT sediment yield results in this area of the UK.
- To adequately represent EDL land cover and use within SWAT ADAS standard crop rotations had to be adjusted. The representation of the Wick series soil within the ADAS soil texture class had to be changed.

The use of data sets with varying sampling frequency for calibration and validation affected the outcome of the calibration and validation results. Total phosphorus calibration was undertaken using Environment Agency monthly measured data;

however validation was partly done using weekly and daily data. When considering phosphorus dynamics on a daily basis SWAT is not able to predict total phosphorus concentrations with great accuracy. Johnes (1996b) daily data clearly show that total phosphorus concentrations within the study area are very dynamic, changing on a daily basis. SWAT is unable to predict such variability for two main reasons.

1: SWAT is unable to predict individual event based sediment or water quality values.

2: Limited data were available for actual STW phosphorus discharge rates.

Consequently although daily data are available for calibration SWAT performs better when comparing predicted total phosphorus values to either monthly or weekly observed data. However SWAT can model flow or nutrient exceedance quite well.

9.2 Future Scenarios

Scenarios are neither predictions nor forecasts of future conditions. Rather they describe alternative plausible futures that conform to sets of circumstances or constraints within which they occur (Hammond, 1996). The true purpose of scenarios is to illuminate uncertainty, as they help in determining the plausible futures (Fisher, 1996).

This analysis has demonstrated that it is possible to integrate modelled future climate change data involving socio-economic scenarios considering future agricultural land use into SWAT within the study area. The model outputs are useful in terms of assessing the impact of varying nutrient loads and hydrological dynamics within the two modelled systems. It has shown that there are considerable challenges to doing so and also that there are considerable advantages.

As it is not possible to predict the future, such studies are based on scenarios. In this study two climate scenarios and two socio-economic scenarios have been used based on the UKCIP02 climate scenarios. The results for the scenarios, which have been chosen to represent a plausible range of potential futures, clearly identify the issues of concern. Although the scenarios are termed 2050s there is nothing in the scenarios which defines the date. Thus a 2050s Low climate scenario could be expected to give similar results to a 2020 High or a 2080 'Very Low' scenario (Audsley *et al.,* 2001). The use of a baseline scenario is used to enable the effects of climate and land use change to be identified.

The results of the RegIS project show that with future socio-economic scenarios a reduction in nutrient loads within the East Anglia region can be expected (Audsley *et al.,* 2001). This is in contrast to this research, where although climate scenarios show the greatest impact on the study area in terms of nutrient loading, socio-economic scenarios also result in an increase in nutrient loading to the system. This is because RegIS output predicted that there would be increased flood risk in the East Anglian region, which makes extensive areas of land no longer suitable for arable agriculture.

The consequences of this increased flood risk within the region mean that in the RegIS project areas of land either have no cropping where the flood risk is very high, or grass where the land is unsuitable for arable but remains suitable for pastoral agriculture. The density of arable crops in the East Anglian region implies that there are few changes to crop distribution between the current and the two future socio-economic scenarios (RE and GS) except those due to increased flooding. Within SWAT it is not possible to model flooding therefore different densities of crops have been used within the study area to those seen in the RegIS system, as all current agricultural land has been modelled as arable or pastoral agricultural. This has resulted in different results to those predicted in RegIS. However when the area is modelled with a greater proportion of grassland, as can be seen with the management solution results, reduction in nutrient loads to system can be seen under the two socio-economic scenarios. This is thus in agreement with RegIS.

9.2.1 Implications for the Broads

The Broads are a very complex system in terms of hydrology and nutrient loads and dynamics. The use of SWAT to represent this complex system has enabled this research to highlight a number of implications for the study area for both the current situation and in terms of future scenarios. This is especially so for current and future watercourse monitoring programmes as well as land management practices and land use.

The implications of this research in the context of the Water Framework Directive are far reaching. The Broads are unique in character and the distinctiveness of the area is the key to its future well-being. One of the main issues therefore is the need to protect the landscape character, and to conserve the Broads as a living, working landscape for future generations. The lakes are an essential part of the Broads, both of its landscape and its functioning. This research shows that in the longer-term, the over-riding issue is climate change. There are also more immediate issues of the protection of water resources and water quality which need to be addressed. Implementation of the Water Framework Directive will be important. In particular catchment management, as has been demonstrated in this work will be critical for the future of the Broads.

The Broads are managed by the Broads Authority (BA) but the lake systems are affected by their wider surrounding catchments which do not fall under the BA's remit. In order to carry the catchment management approach forward all agencies and bodies such as local farmers, the Broads Authority, the Environment Agency, English Nature and DEFRA need to work together to pool resources and information. This is starting to be accomplished, English Nature has started work in partnership with the Environment Agency, the Countryside Agency and the Rural Development Service in four pilot river catchment areas to find ways to reduce diffuse pollution to deliver the Water Framework Directive target of 'good ecological quality'.

The EA is the principal protector of the water environment in England and Wales. They are already responsible for a wide range of work required under the Water Framework Directive, but some of their activities will need to change. Results of this research indicate that water monitoring programmes must be reviewed and diffuse water pollution addressed by working with other relevant authorities.

If agriculture is to continue to be an essential part of the Broads economy then changes in agricultural policy and farming practices must be achieved with maximum support from all relevant authorities so that agriculture's impact on water quality is reduced. English Nature and the EA need to be encouraged to make farmers aware of the likelihood of longer term regulation, if management practices do not change in the near future. Many issues of diffuse pollution will require a significant promotional effort to farmers, if they are to be taken seriously. At local level, a mix of detailed technical advice and demonstration, flexible grant aid, and the potential threat of action via a regulatory approach is maybe an effective combination of policy instruments to tackle issues in priority areas, such as Hickling Broad.

Current Situation

The first Chapters of this thesis show that a considerable amount of work has already been done in the Broads and that there is already much known about the study area. A lot of work has been based on EA monitored data. EA sampling is either undertaken at a monthly or bi-monthly frequency. Published recommended sampling frequencies suggest that this is adequate, for small river basins (10000 km^2 or less). A maximum of 24 samples per year are needed to assess river trends and for eutrophic lakes 12 samples per year are required, including bi-monthly samples during the summer (UNEP/WHO, 1996).

This research shows, along with work carried out by Johnes (1996b) that current water quality monitoring schemes operating on a monthly or bi-monthly basis systematically underestimate nutrient loads, especially phosphorus which is largely exported in particulate form from agricultural sources.

Severa-Martinez (2005) defined an ecological failure criterion for Hickling Broad. In terms of total phosphorus a maximum ecological threshold limit of 0.1 – 0.25 mg l^{-1} was suggested; total phosphorus concentrations above these values will have an adverse effect on Hickling Broad. If only EA monthly data are considered then no adverse affects on Hickling Broad would be expected. However if SWAT results are used then both thresholds are breached 20% of the time, having an adverse effect on Hickling Broad. This would be underestimated or missed if only monthly data were considered.

Johnes (1996b) data and SWAT results show that it is at higher flows that higher phosphorus concentrations are likely. EA data are predominantly collected at lower flows as discussed in Chapter 6. Work carried out by USGS (2001) suggests that the best overall monitoring strategy for accurate and precise load and trend estimations of sediment and nutrients consists of 50% base flow samples and 50% storm samples. Analysis of EA data showed that samples were nearly always taken at low flows. Therefore based on monthly samples the ability to ensure adequate sampling of all river stages is severely limited.

SWAT results, simple statistical calculations and other studies therefore suggest that it would be advisable to undertake weekly, daily, sub-daily or event based sampling. This would give a more detailed knowledge of nutrient dynamics of this complex system and could be used to further calibrate and validate the SWAT model.

Future Scenarios

The results of future scenarios run with SWAT have been discussed in chapter 8, the following points summarise the findings:

- An increase in rainfall and temperature through climate change increase eutrophication problems for both the rivers and Broads within the Bure and Ant system. This will increase the already high risk of ecological failure in Hickling Broad.
- For both the future climate scenarios there are increases in rainfall, run-off, leaching and infiltration affecting the mobilisation of N. Where infiltration values increase there is an increase in N fluxes to the catchment systems, suggesting the main transfer route for N in the study area is through groundwater.
- For the climate only scenarios the spatial distribution for each output parameter does vary depending on the climate input, as this is the only factor, which is being changed. A spatial relationship between rainfall and run-off cannot be clearly seen. This may be caused by the lack of spatial relationship between the MONARCH grid cells used in the baseline and future scenarios.
- Spatial results do however show that higher P outputs are linked to areas of higher run-off and consequently higher sediment loading; no matter how low

the P inputs to the system. Therefore high P source areas can be identified and should be the focus for P reduction activities.

- The effect of the two future socio-economic scenarios is very different with the GS scenario decreasing the nutrient loads to the two catchment systems below that of current conditions.
- Both future socio-economic scenarios also show a substantial increase in potential demand for irrigation due to the increased area of sugar beet and potatoes; this is however assuming unlimited water supplies.
- It is climate change that causes the greatest increases in nutrient, and sediment yield and changes in hydrological dynamics in the study area. To alleviate increased yields and increased river flow to the system erosion control measures can be employed.
- Modelling results suggest that the only sustainable future scenario in terms of land management would be to revert all the agricultural land to grassland. However the feasibility of this in terms of agricultural economics is questionable and needs to be investigated further.

Eutrophication problems in the study area, both now and in the future, need to be managed and reduced to reduce the risk of further ecological failure in the Broads. The use of SWAT within this research has shown that this is possible through managing the Broads, particularly the Thurne river basin as a whole catchment, by reducing nutrient loads to the system through erosion control practices. Eutrophication problems within the Broadland system need to be dealt with at their source, by reducing nutrient loads to the catchment and changing land use cover and practices instead of concentrating on restoring the Broads through on-site measures such as dredging and bio-manipulation. Once water quality has been improved on a catchment wide-basis restoration measures such as the use of bio-manipulation can be attempted in the Broads.

9.3 Conclusions

In summary, the important conclusions that can be drawn from the results of this study are:

1: Based on the current available data, the SWAT model demonstrated its utility as a tool to understand processes in the catchment and as a basis for effective management in the Bure, Ant and Thurne river basins.

2: The use of UK soil associations instead of soil series within SWAT had little impact on predicted sediment yields. Therefore the use of the National soil map and soil associations instead of more detailed soil data has a limited influence on SWAT sediment yield results in this area of the UK. This finding also proves significant for other hydrological or related studies using the National soil map data sets.

3: Investigation into the use of ADAS standard crop rotations showed that these may not represent crops grown when compared to census data such as EDL.

4: On the basis of the nutrient calibration results it has been concluded that SWAT has a shortcoming in nutrient representation due to the sensitivity of the model to many parameters.

5: The use of SWAT within such a hydrologically complex study area indicated limitations in the predictive capability of the model, especially for sediment.

6: It is possible to integrate modelled future climate change data involving socio-economic scenarios considering agricultural land use into SWAT within the study area.

7: This research shows, along with work carried out by Johnes (1996b) that current water quality monitoring schemes operating on a monthly or bi-monthly basis systematically underestimate nutrient loads and the consequent risk of ecological failure of Hickling Broad.

8: Increased rainfall and higher temperatures through climate change will increase nutrient and sediment loads and alter the hydrological dynamics of the study area, threatening the ecological condition of Hickling Broad.

9: Due to increased water demand irrigation modelling using CropWat suggests that under future climate and economic scenarios current agricultural practices are not sustainable.

10: Erosion control measures, particularly critical area planting in the north of the Thurne system, should be employed in the current situation and future scenarios if the risk of ecological failure to Hickling Broad is to be minimised

9.4 Future Work

Due to the fixed time limit set for this research the following ideas were not explored in full. Therefore this research could continue in the following directions:

- The under estimation of sediment loading and the modelling of all sediment sources with the SWAT model needs to be investigated.
- A better way to represent irrigation depth within SWAT needs to be addressed.
- The increased flood risk within the study area and consequent removal of agricultural land from production should be investigated in SWAT.
- Modelling other erosion control measures with the existing input variables and future scenarios is an interesting area for further research.
- Investigation into a more rigorous water quality sampling regime which is suitable for the complex system of the Broads should be undertaken.
- Research should also be undertaken to investigate whether there are other models possible for modelling of low lying pumped systems.
- Leading on from this work a socio economic analysis investigating the feasibility of current agricultural practices under future scenario conditions should be considered.

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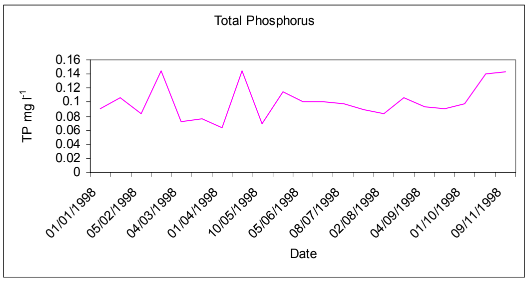
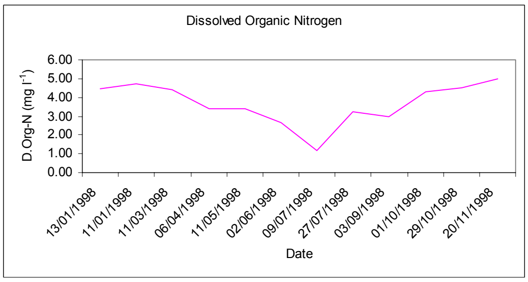
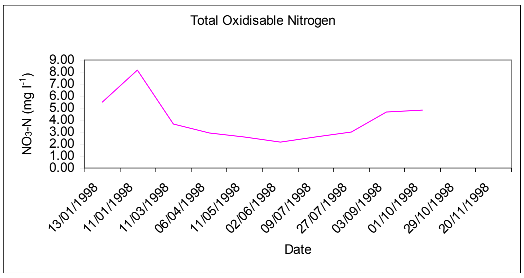
APPENDIX ONE
1998 Nutrient Concentrations for the Bure, Ant and Thurne River Basins

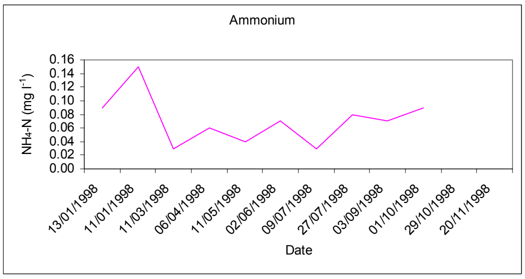
Nitrogen and Phosphorus Concentrations at Ingworth, River Bure (1998)

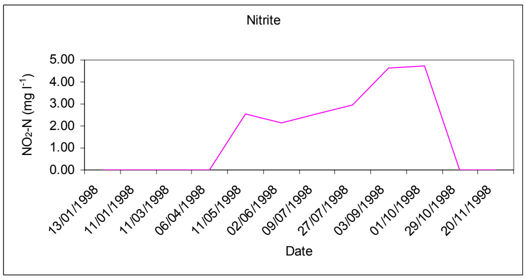


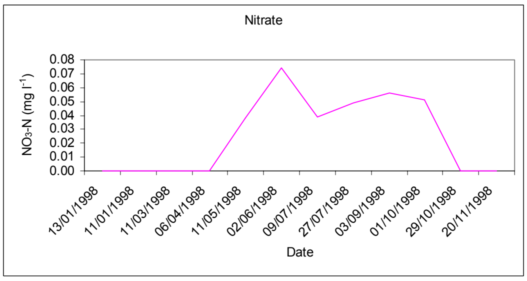


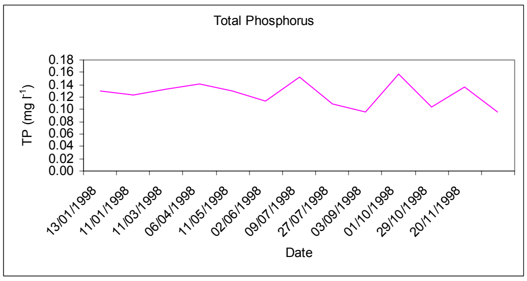
Nitrogen and Phosphorus Concentrations at Scarrow Beck, River Bure (1998)



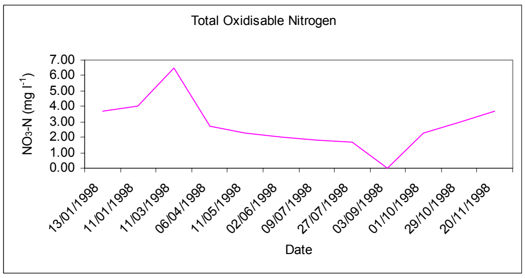


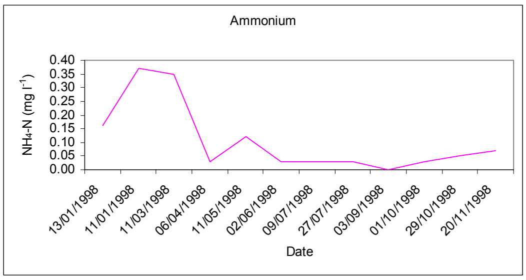


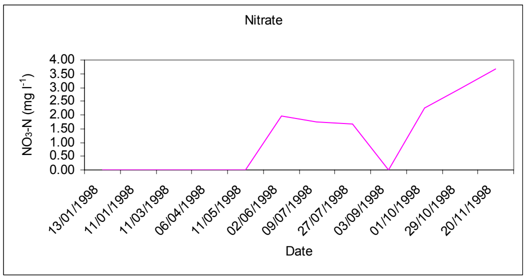


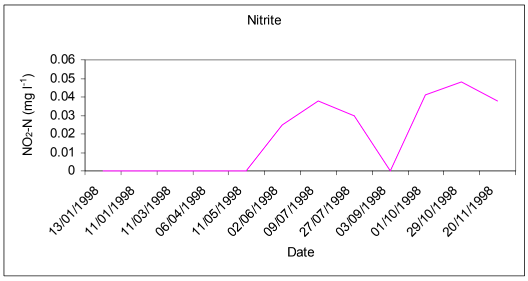


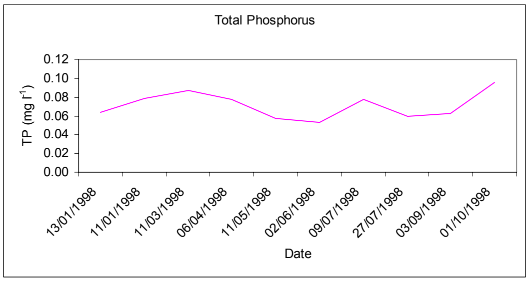
Nitrogen and Phosphorus Concentrations at Kings Beck, River Bure (1998)



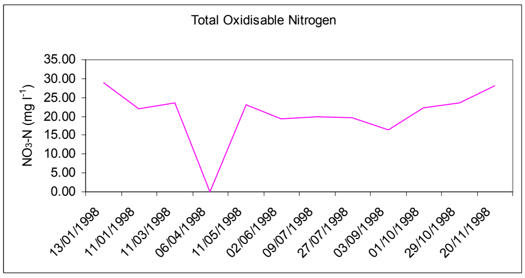


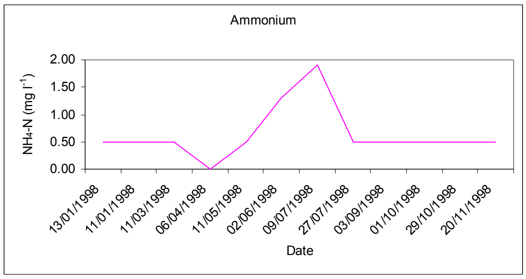




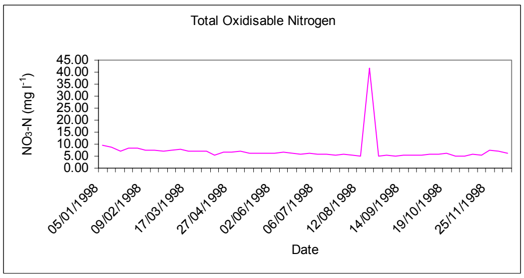


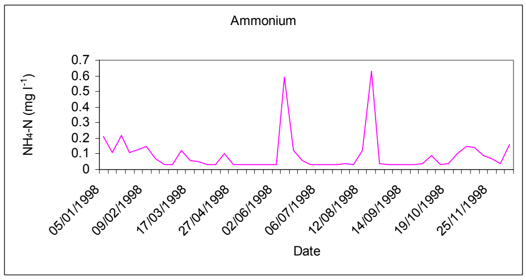
Nitrogen and Phosphorus Concentrations at Aylsham, River Bure (1998)

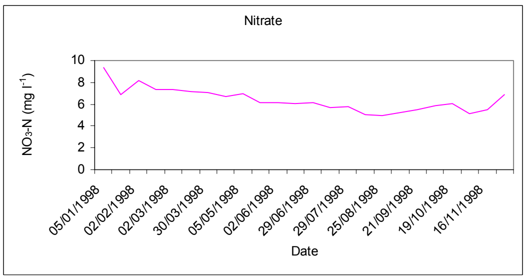


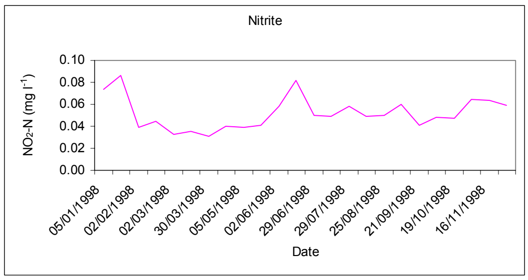


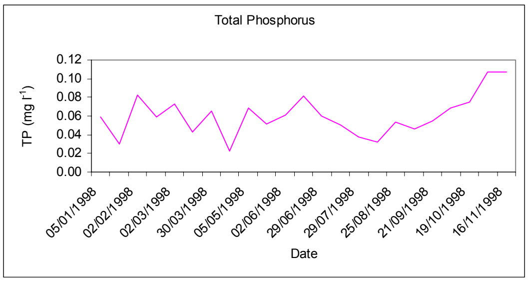
Nitrogen and Phosphorus Concentrations at Horstead Mill, River Bure (1998)



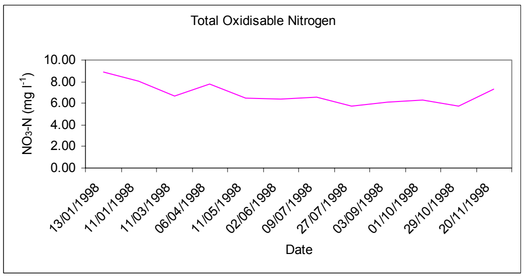


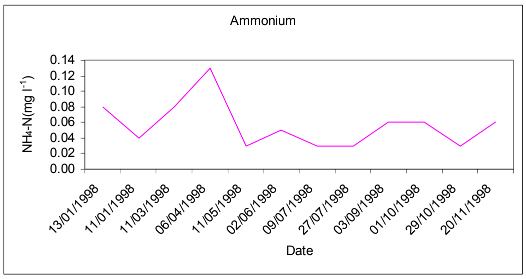


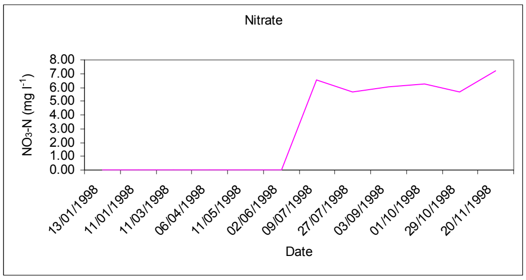


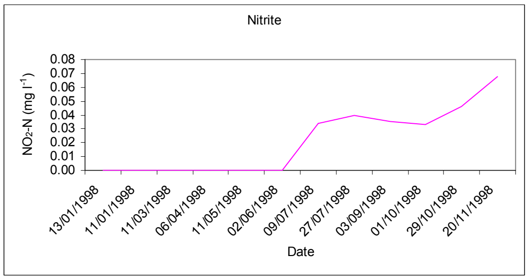


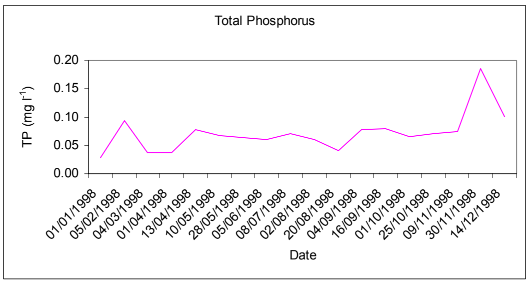
Nitrogen and Phosphorus Concentrations at Wroxham Rail Bridge, River Bure (1998)



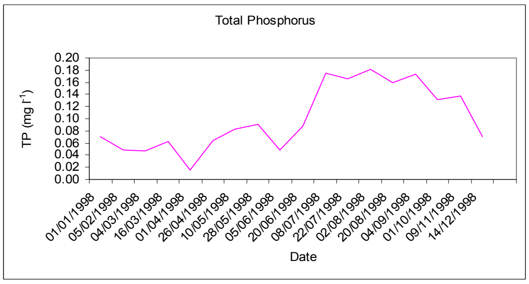




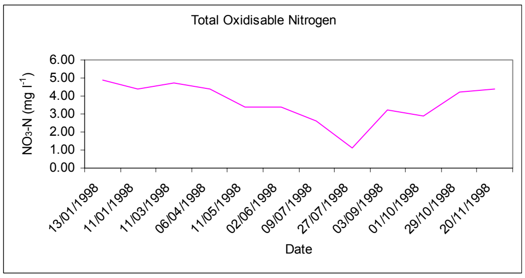


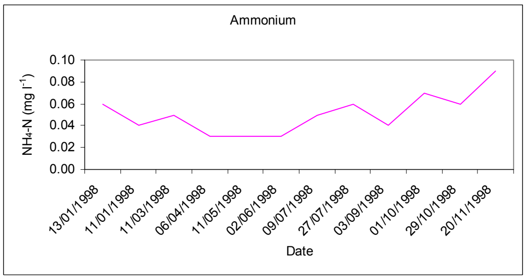


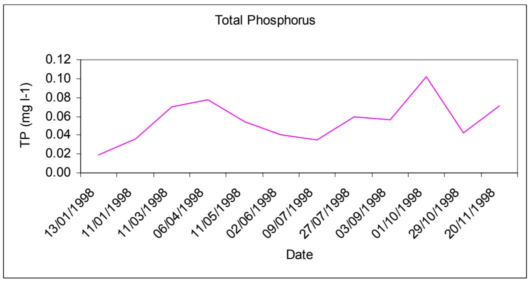
Nitrogen and Phosphorus Concentrations at St Benets Abbey, River Bure (1998)



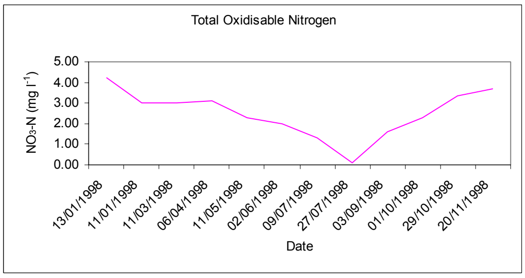
Nitrogen and Phosphorus Concentrations at Honing Lock, River Ant (1998)

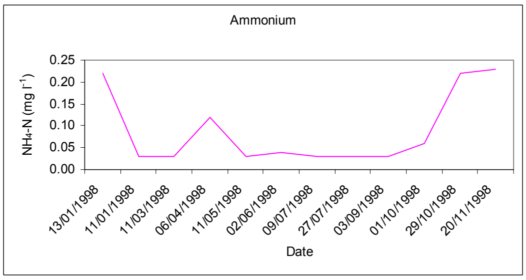


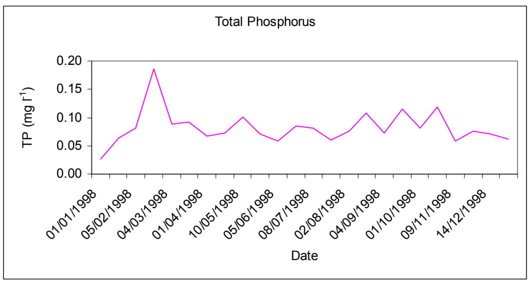




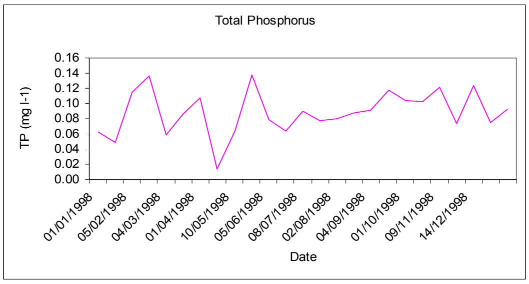
Nitrogen and Phosphorus Concentrations at Wayford Bridge, River Ant (1998)



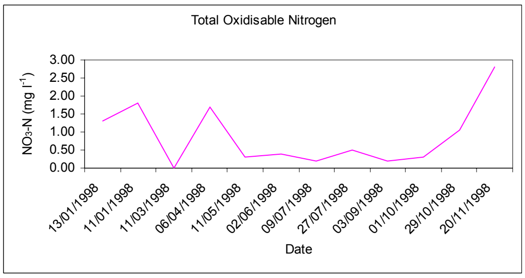


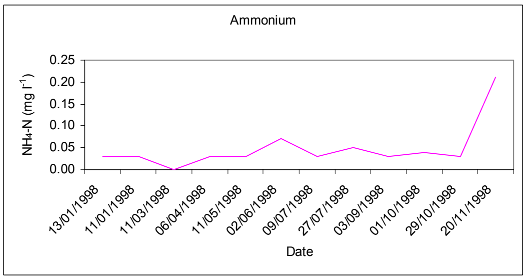


Nitrogen and Phosphorus Concentrations at Hunsett Mill, River Ant (1998)

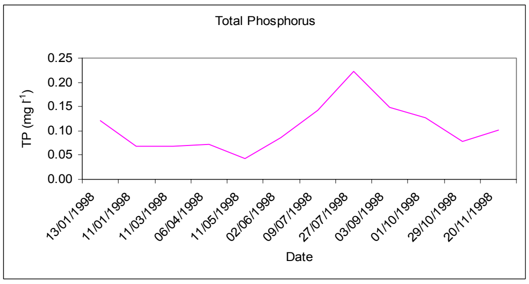


Nitrogen and Phosphorus Concentrations at Irstead Church, River Ant (1998)

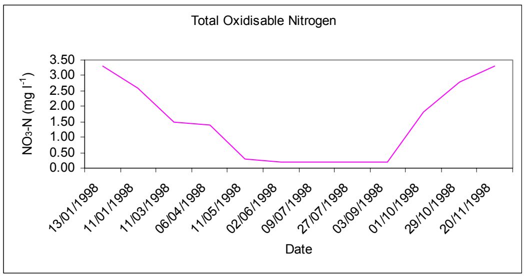


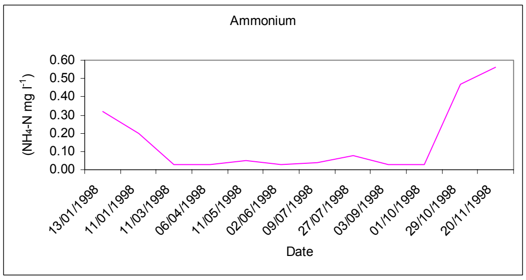


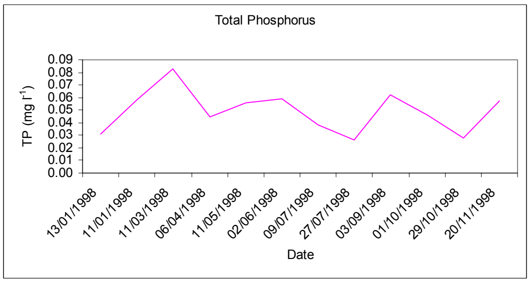
Nitrogen and Phosphorus Concentrations at How Hill, River Ant (1998)



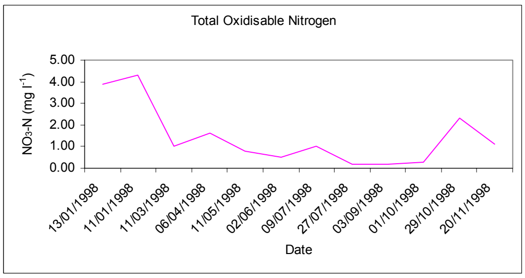
Nitrogen and Phosphorus Concentrations at Martham Ferry, River Thurne (1998)

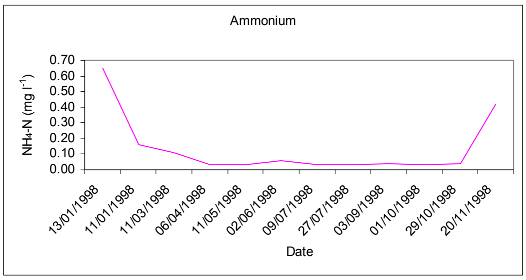




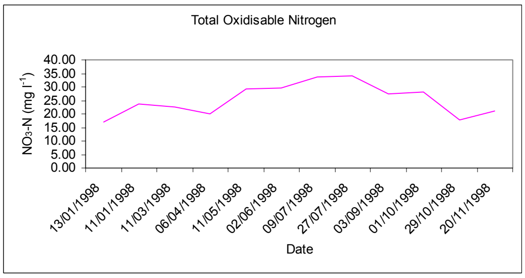


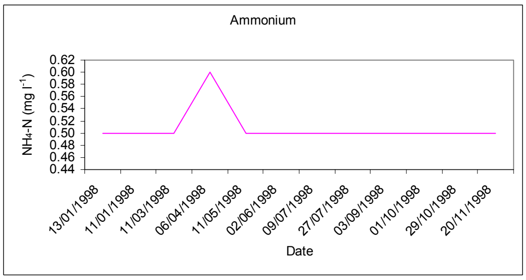
Nitrogen and Phosphorus Concentrations at Potter Heigham, River Thurne (1998)



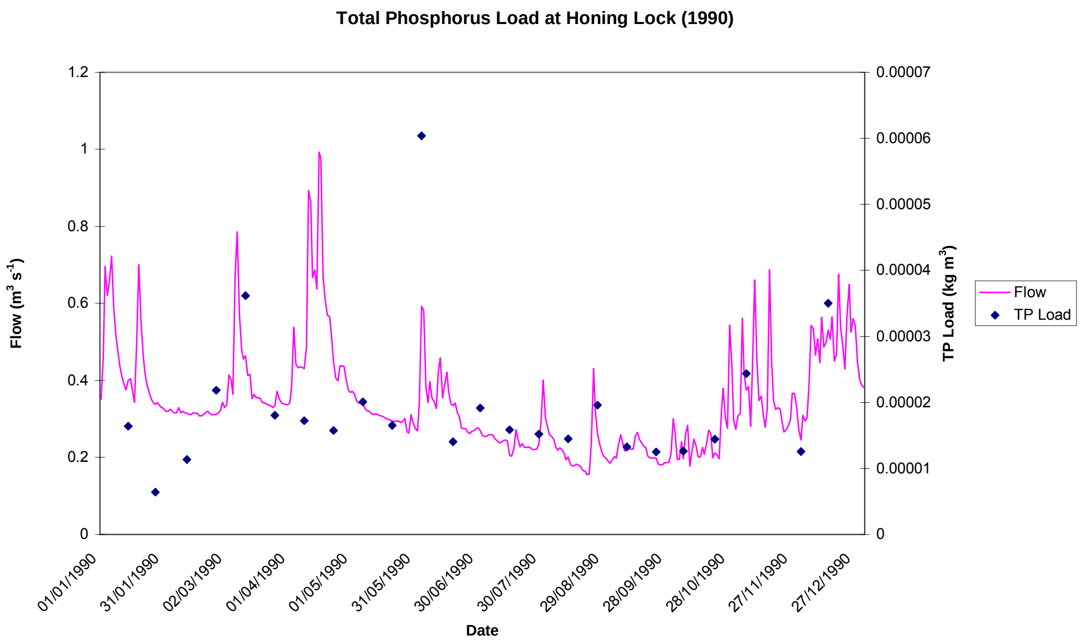


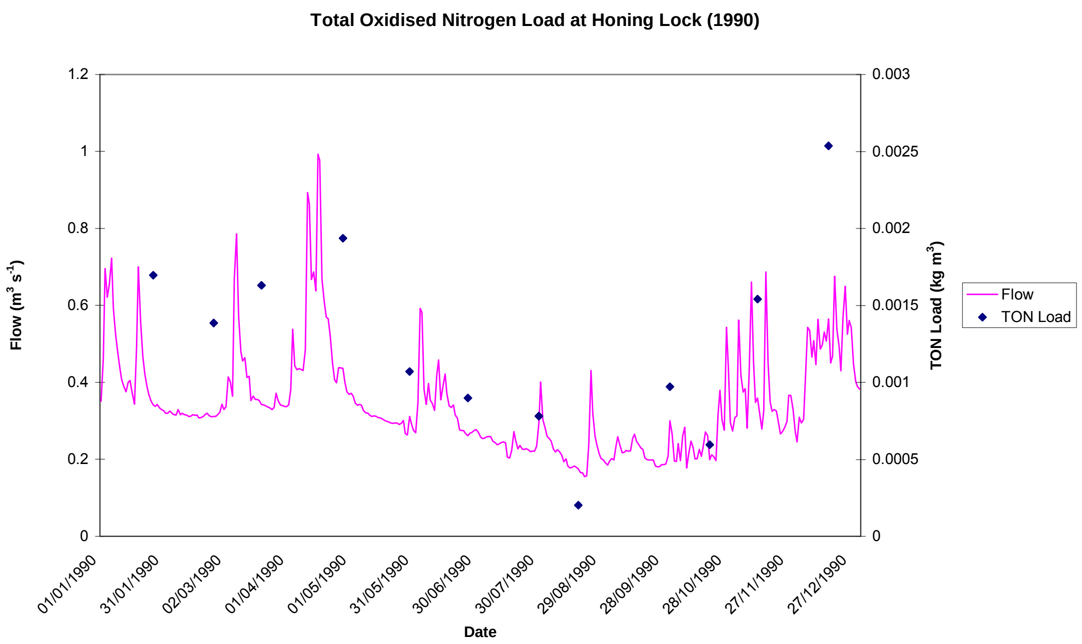
Nitrogen and Phosphorus Concentrations at Ludham STW, River Thurne (1998)

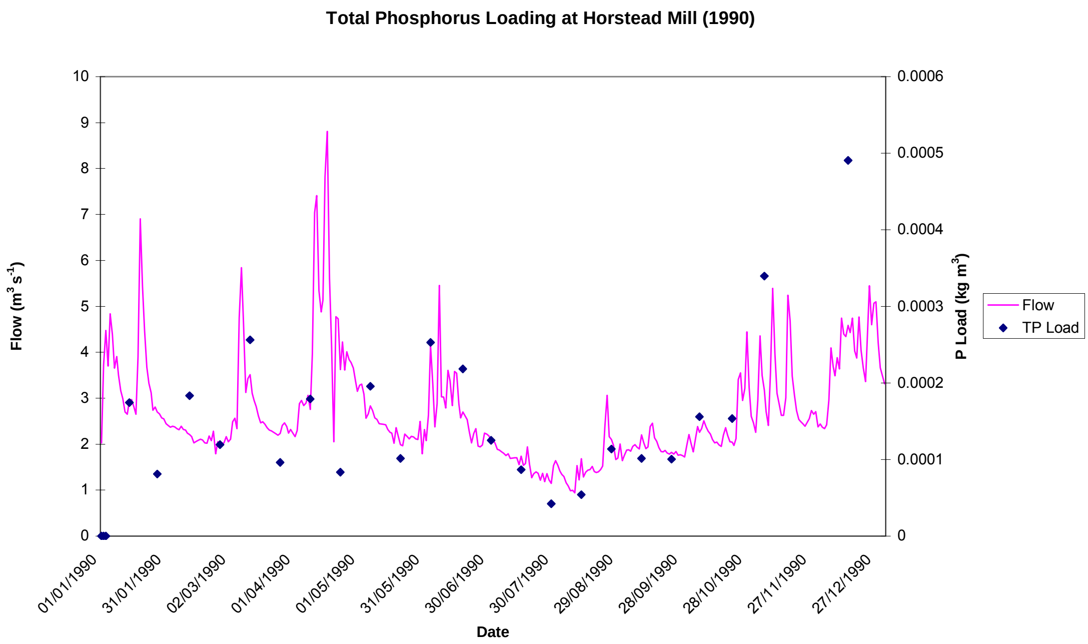




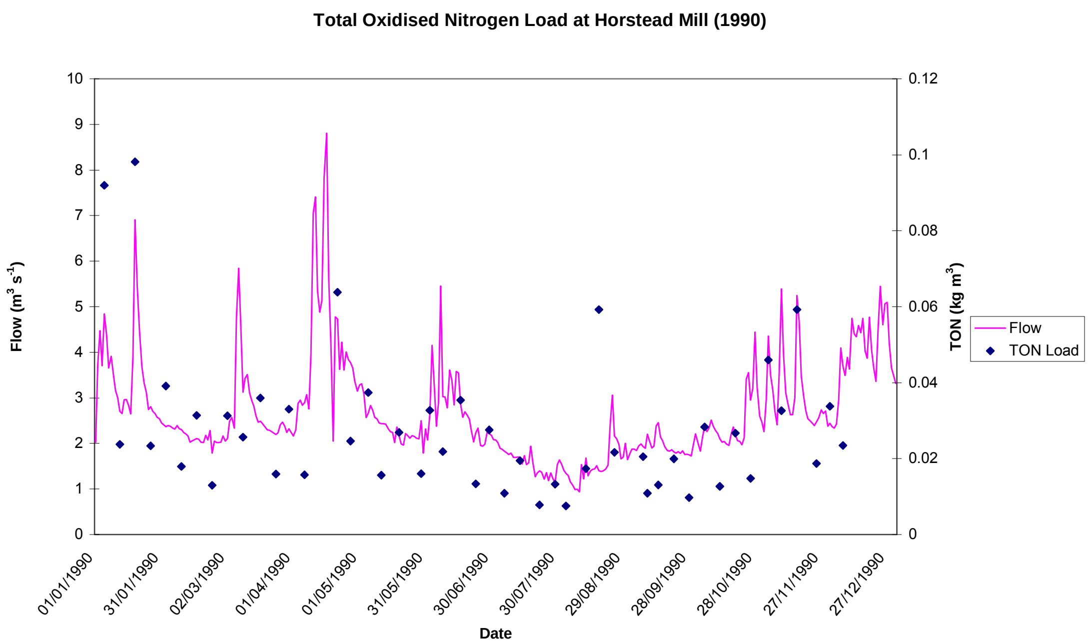
APPENDIX TWO
1990 Nutrient Loads for Horstead Mill and Honing Lock


Total Phosphorus Load at Honing Lock (1990)
Flow (m3 s-1)
TP Load (kg m3)
Flow
TP Load
Date
1.2
1
0.8
0.6
0.4
0.2
0
0.00007
0.00006
0.00005
0.00004
0.00003
0.00002
0.00001
01/01/1990
31/01/1990
02/03/1990
01/04/1990
01/05/1990
31/05/1990
30/06/1990
30/07/1990
29/08/1990
28/09/1990
28/10/1990
27/11/1990
27/12/1990


Total Oxidised Nitrogen Load at Honing Lock (1990)
Flow (m3 s-1)
TON Load (kg m3)
Flow
TON Load
Date
1.2
1
0.8
0.6
0.4
0.2
0
0.003
0.0025
0.002
0.0015
0.001
0.0005
01/01/1990
31/01/1990
02/03/1990
01/04/1990
01/05/1990
31/05/1990
30/06/1990
30/07/1990
29/08/1990
28/09/1990
28/10/1990
27/11/1990
27/12/1990



Total Phosphorus Loading at Horstead Mill (1990)
Flow (m^3 s^{-1})
P Load (kg m^3)
Date
Flow
TP Load
0
1
2
3
4
5
6
7
8
9
10
0
0.0001
0.0002
0.0003
0.0004
0.0005
0.0006
01/01/1990
31/01/1990
02/03/1990
01/04/1990
01/05/1990
31/05/1990
30/06/1990
30/07/1990
29/08/1990
28/09/1990
28/10/1990
27/11/1990
27/12/1990

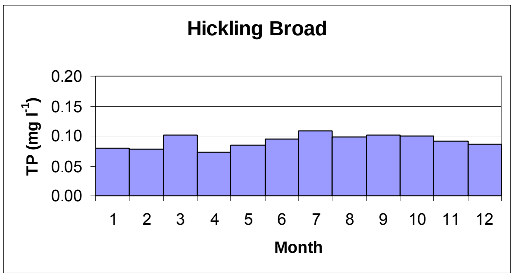


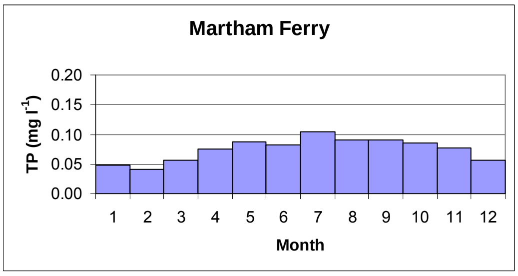
Total Oxidised Nitrogen Load at Horstead Mill (1990)
Flow (m^3 s^{-1})
TON (kg m^3)
Date
Flow
TON Load
0
1
2
3
4
5
6
7
8
9
10
0
0.02
0.04
0.06
0.08
0.1
0.12
01/01/1990
31/01/1990
02/03/1990
01/04/1990
01/05/1990
31/05/1990
30/06/1990
30/07/1990
29/08/1990
28/09/1990
28/10/1990
27/11/1990
27/12/1990

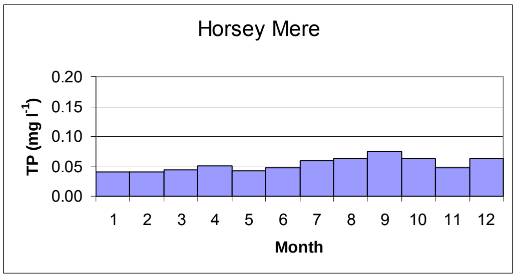
APPENDIX THREE

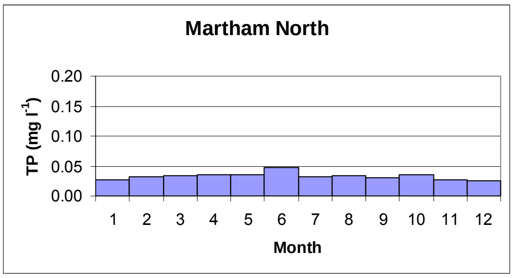
Monthly Mean Total Phosphorus and Total Oxidised Nitrogen Concentrations in the Upper Thurne Broads (1978 – 2001)

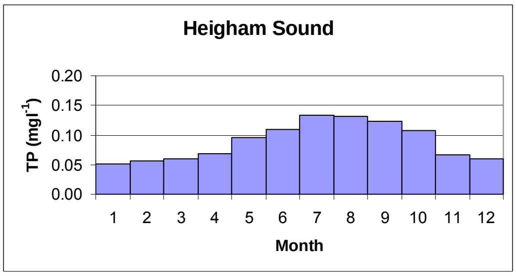
Monthly mean total phosphorus concentrations for the Upper Thurne Broads (1978 - 2001)

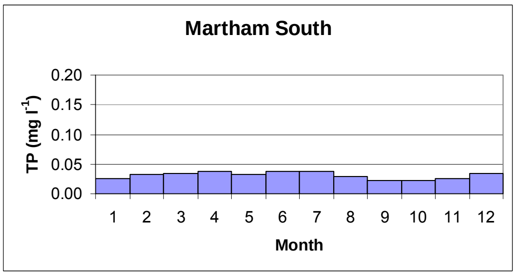


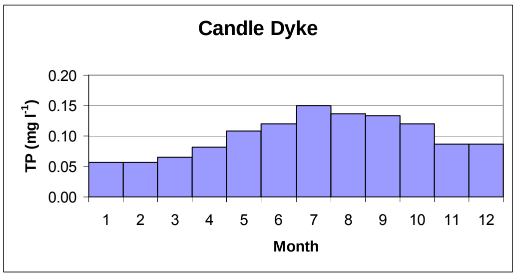




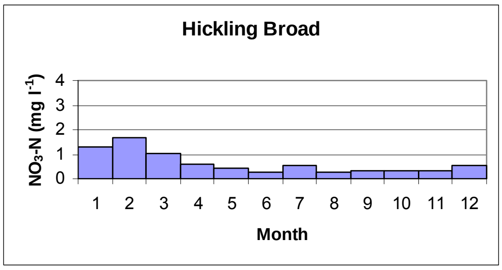


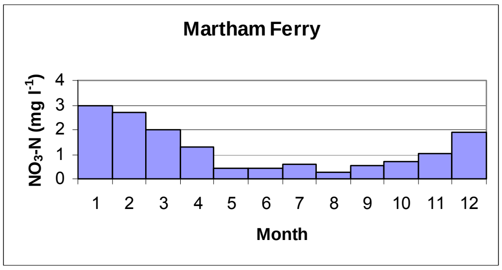


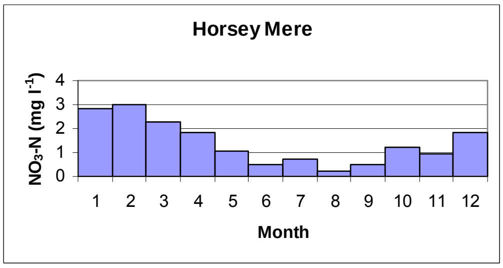


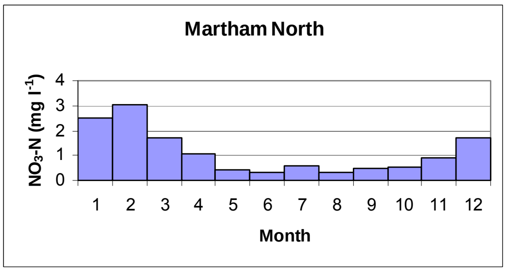


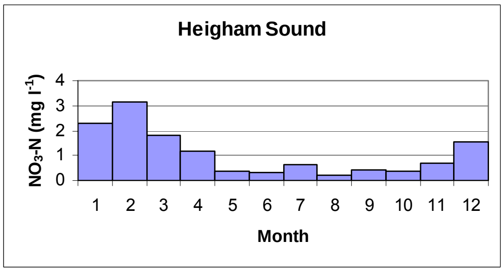
Monthly mean total oxidised nitrogen concentrations for the Upper Thurne Broads (1978 -2001)

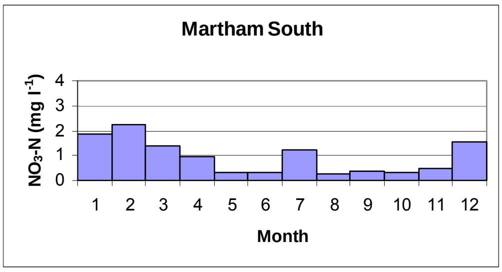


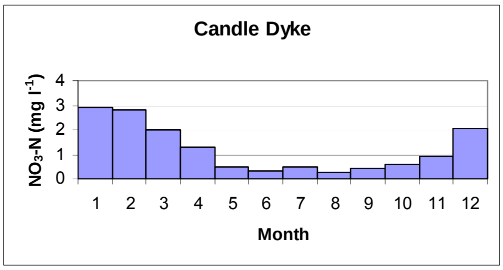




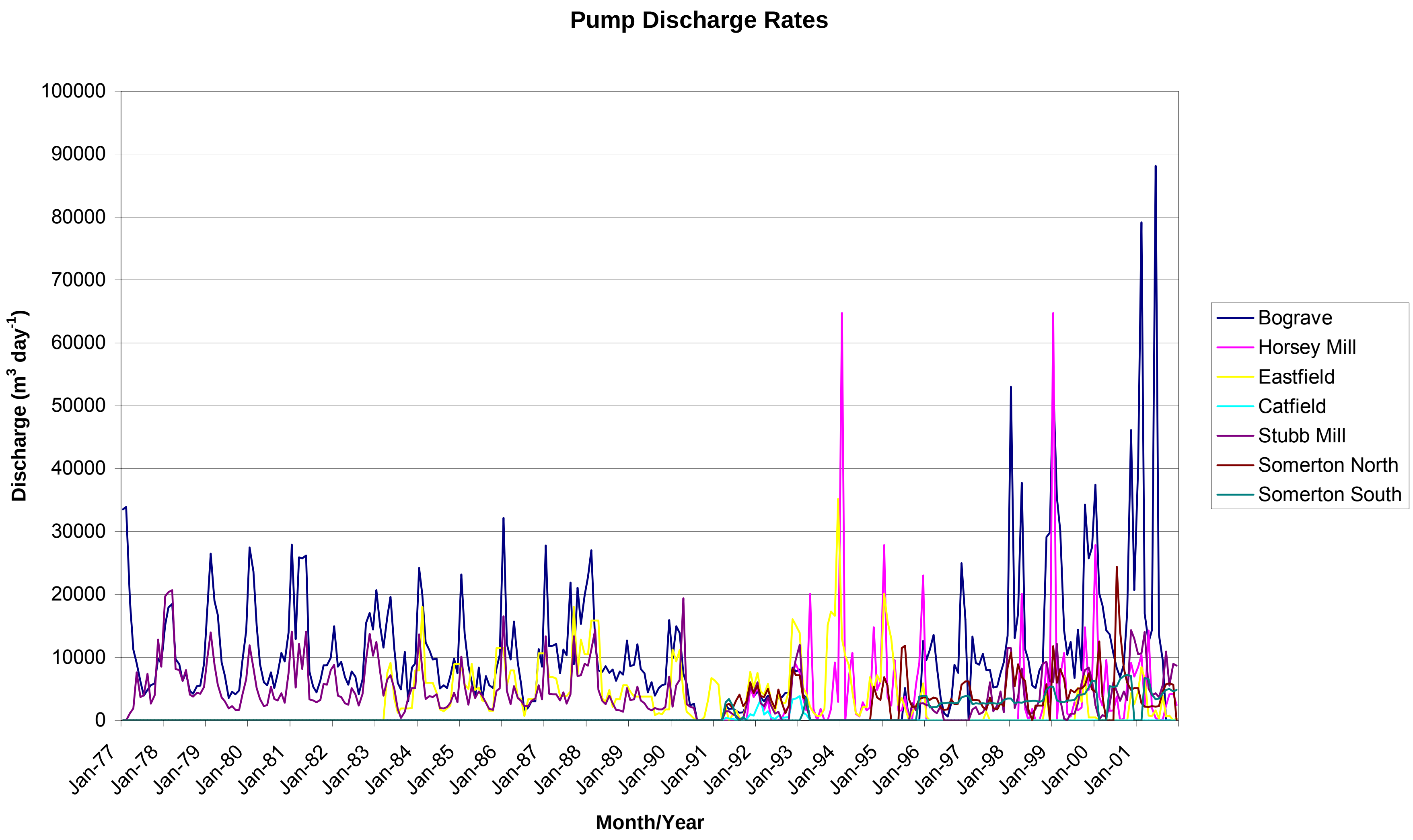








APPENDIX FOUR
Upper Thurne Internal Drainage Board Pump Discharge Rates (1977 – 2001)

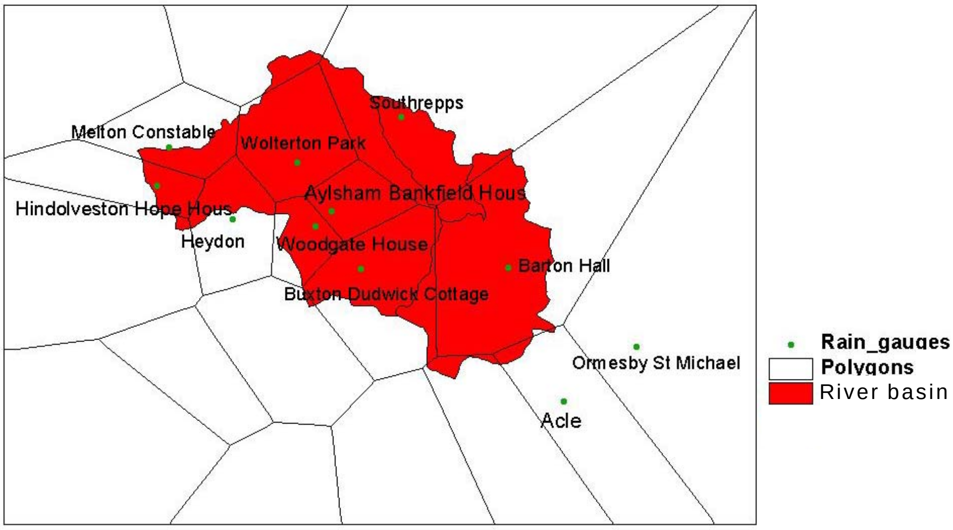

Pump Discharge Rates
Discharge (m3 day-1)
100000
90000
80000
70000
60000
50000
40000
30000
20000
10000
0
Jan-77
Jan-78
Jan-79
Jan-80
Jan-81
Jan-82
Jan-83
Jan-84
Jan-85
Jan-86
Jan-87
Jan-88
Jan-89
Jan-90
Jan-91
Jan-92
Jan-93
Jan-94
Jan-95
Jan-96
Jan-97
Jan-98
Jan-99
Jan-00
Jan-01
Month/Year
Bograve
Horsey Mill
Eastfield
Catfield
Stubb Mill
Somerton North
Somerton South

APPENDIX FIVE
SWAT Data Collection

	Data Required	Source	Obtained	Years	Details
Temporal Data					
Weather	Rainfall	BADC	YES	90 - 99 (for most gauges)	Daily Rainfall for 16 gauges (drain files). Hourly rainfall for Hemsby and Coltishall (hrain files)
	Temp. (max, min, wet and dry bulb)	BADC	YES	90 - 2000	Three stations. Max and Min in temp files. Wet and dry bulb in hwx files
	Solar radiation	BADC	YES	90 - 2000	Hourly and daily radiation for 1 site (Hemsby)
	Wind speed	BADC	YES	90 - 2000	Hourly mean for 2 sites (Hemsby and coltishall). Hwnd files
	Soil temp.	BADC	YES	90 - 2000	Daily soil temperature for 2 sites (Hemsby and Melton Constable). Tsoil files.
	Sunshine duration	BADC	YES	90 - 2000	Hourly sunshine for Coltishall (hsun files)
Land Management Practices	Agrochemicals	DEFRA	NO	-	Hough
	Timing	DEFRA	YES	-	Hough
	Amount applied	DEFRA	YES	-	ADAS Best Practice
	Cropping regime	ADAS	YES	-	Standard rotations for the Eastern region only
Abstractions	Ground water	EA	Yes	-	
	Surface water	EA	Yes	-	
Discharges	Sewage and domestic outfalls	EA	Yes	-	
	Industrial discharge	EA	Yes	-	
	Farming activities e.g. sheep dips	EA	NO	-	
	Fish farms	EA	NO	-	
Spatial Data					
	DEM	CEH	YES	-	Problems with importing into ArcView
	River network	CEH	YES	-	Problems with importing into ArcView
	Land use	CEH	YES	1990	Needs to be combined with NUTs data and ward boundaries
	Geology		NO		
Soil	Spatial variability	NSRI	YES	-	-
	Soil type	NSRI	YES	-	-
	Properties	NSRI	YES	-	-
Digital Boundaries	Wards	Digimap	YES	1991	Needs to be combined with NUTs data and ward boundaries
	Catchment	Anglian Water	YES	-	-
Validation Data					
	Water quality data	EA	YES	Vary to 2000	9 sites on the Bure, 6 on the Ant and 4 on the Thurne
	Flow data	EA	YES	Honing Lock (Ant) 1966 - present Horstead Mill (Bure) 1974 - present	No flow data for the Thurne, but do have pump data

APPENDIX SIX
Areal Average Rainfall Calculations using Thiessen Polygons Method

Thiessen Polygons



Areal Average Rainfall

Rain Gauge Name	ArcView Area	Proportion of Catchment	Annual Rainfall (mm)	Weighted Rainfall	Near TP mean	Near A mean
Acle	73780912	9.37	529.70	49.62	32.00	-61.62
Attle Bridge	448202	0.06	626.30	0.36	-64.60	-158.22
Aylsham Bankfield House	205104048	26.04	552.80	143.96	8.90	-84.72
Barton Hall	40165776	5.10	569.20	29.03	-7.50	-101.12
Buxton Dudwick Cottage	82675992	10.50	606.50	63.67	-44.80	-138.42
Heydon	24202898	3.07	606.80	18.65	-45.10	-138.72
Hindolveston Hope House	17169578	2.18	655.50	14.29	-93.80	-187.42
Melton Constable	21375778	2.71	625.00	16.96	-63.30	-156.92
Ormesby St Michael w.wks	127116928	16.14	516.70	83.39	45.00	-48.62
Southrepps	84848048	10.77	536.50	57.80	25.20	-68.42
Wolterton Park	84848048	10.77	596.90	64.30	-35.20	-128.82
Woodgate House	25857796	3.28	599.30	19.68	-37.60	-131.22
Total	**787594004**	**100.00**	**468.08**	**561.70**		
	Thiessen Mean	**468.08**	**Arithmetic Mean**	**468.08**		

APPENDIX SEVEN
USLE Calculations

USLE Calculations: Wischmeier W and Smith D (1978) *Predicting rainfall erosion losses: a guide to conservation planning*. Agriculture Handbook 282. USDA-ARS

E = R x K x L x S x C x P

Where E is the mean annual soil loss, R is the rainfall erosivity factor, K is the soil erodibility factor, L is the slope length, S is the slope steepness factor, C is the crop management factor and P is the erosion control practice factor.

Table 1: R, LS and P factors

Factor	Calculation	Data Source	Value
R	11.54 x exp(0.00215 x annual rainfall)	Min and Max rainfall taken from Aylsham Bankfield Hoise (1990 – 1999)	Min = 517.4mm Max = 778.3mm
LS	(Slope length/22)^n x (0.065 + (0.045 X slope %) + (0.0065 x slope %^2))	Taken from Morgan (1995) Soil Erosion & Conservation (2nd Ed), Longman. n = 0.4 for 3 degrees, 0.3 for 2 degrees and 0.2 for 1 degree. Max and min values taken from LS calculations for each HRU in river basin (table 2)	Min = 0.18 Max = 0.49
P	No erosion control practices	Taken from Morgan (1995) Soil Erosion & Conservation (2nd Ed), Longman	1

Table 2: LS calculations

	Slope Length	Slope Steepness		Slope
HRU	(m)	(m/m)	(%)	LS Factor
1	60.98	0.05	3.24	0.42
2	60.98	0.05	3.24	0.42
3	91.46	0.05	3.05	0.47
4	91.46	0.05	3.05	0.47
5	91.46	0.05	3.05	0.47
6	60.98	0.06	3.56	0.46
7	60.98	0.06	3.56	0.46
8	60.98	0.05	3.24	0.42
9	60.98	0.05	3.24	0.42
10	91.46	0.05	2.99	0.46
11	91.46	0.05	2.99	0.46
12	91.46	0.05	2.99	0.46
13	91.46	0.05	2.93	0.45
14	91.46	0.05	2.93	0.45
15	91.46	0.05	2.93	0.45
16	91.46	0.05	2.93	0.45
17	91.46	0.05	2.86	0.44
18	91.46	0.05	2.86	0.44
19	91.46	0.05	2.86	0.44
20	60.98	0.05	3.43	0.45
21	60.98	0.05	3.43	0.45
22	60.98	0.06	3.56	0.46
23	60.98	0.06	3.56	0.46
24	60.98	0.06	3.56	0.46
25	91.46	0.04	2.29	0.36
26	91.46	0.04	2.29	0.36
27	91.46	0.04	2.29	0.36
28	91.46	0.04	2.29	0.36
29	91.46	0.04	2.67	0.41
30	91.46	0.04	2.67	0.41
31	91.46	0.04	2.67	0.41
32	91.46	0.04	2.67	0.41
33	91.46	0.04	2.61	0.40
34	91.46	0.04	2.61	0.40
35	91.46	0.04	2.61	0.40
36	91.46	0.05	3.12	0.47
37	91.46	0.05	3.12	0.47
38	91.46	0.05	2.86	0.44
39	91.46	0.05	3.05	0.47
40	91.46	0.05	3.05	0.47
41	60.98	0.06	3.50	0.45
42	60.98	0.06	3.50	0.45
43	60.98	0.06	3.50	0.45
44	91.46	0.04	2.80	0.43
45	60.98	0.05	3.18	0.41
46	60.98	0.05	3.18	0.41
47	60.98	0.05	3.18	0.41
48	60.98	0.05	3.37	0.44
49	60.98	0.05	3.37	0.44
50	91.46	0.04	2.80	0.43
51	60.98	0.06	3.50	0.45
52	60.98	0.06	3.50	0.45
53	91.46	0.04	2.74	0.42
54	60.98	0.05	3.24	0.42
55	60.98	0.05	3.24	0.42
56	60.98	0.05	3.24	0.42
57	91.46	0.04	2.23	0.35
58	91.46	0.04	2.23	0.30
59	91.46	0.04	2.23	0.30
60	91.46	0.04	2.23	0.30
61	60.98	0.05	3.24	0.42
62	121.95	0.02	1.21	0.18
63	121.95	0.02	1.21	0.18
64	60.98	0.06	3.75	0.49
65	60.98	0.06	3.75	0.49
66	60.98	0.06	3.75	0.49
67	60.98	0.06	3.75	0.49
68	60.98	0.06	3.75	0.49
69	91.46	0.05	3.12	0.47
70	91.46	0.05	3.12	0.47
71	91.46	0.05	3.12	0.47
72	91.46	0.04	2.74	0.42

Table 3: C Factor Values (Wischmeier and Smith 1978)

Crop	C Factor
Wheat	0.5
Winter Barley	0.5
Spring Barley	0.5
Oats	0.5
Other Cereals	0.5
Potatoes	0.64
Sugar Beet	0.64
Fodder 1	0.36
Fodder 2	0.36
Field Beans	0.12
Peas for harvesting dry	0.12
Maize	0.68
Oilseed Rape	0.4
Other Arable	0.4
Linseed	0.4
Bare Fallow	1
All other vegetables	0.45
Orchards	0.006
Small Fruits	0.45
Pasture	0.003
Woodland	0.001

Table 4: USLE K Factor Calculations

									forgc			K factor (t ha^{-1})	
Soils		SAND	SILT	CLAY	OC_ag	OC_pg	fcsand	fcl-si	Arable	Grass	fhisand	Arable	Grass
1024	Adventurers	95	4	1	20.5	18.3	0.20	0.94	0.75	0.75	0.44	0.06	0.06
1022	Altcar	33	33	33	32.1	39.5	0.20	0.81	0.75	0.75	1.00	0.12	0.12
543	Aylsham	47	39	14	1.2	2.6	0.20	0.91	0.87	0.75	1.00	0.16	0.14
711	Beccles	46	29	25	1.7	3.3	0.20	0.83	0.78	0.75	1.00	0.13	0.12
821	Blackwood	70	19	11	2.3	4	0.20	0.87	0.75	0.75	0.95	0.13	0.12
643	Felthorpe	74	19	7			0.20	0.91	1.00	1.00	0.90	0.16	0.16
711	Gresham	41	47	12	1.4	2.5	0.20	0.93	0.83	0.75	1.00	0.16	0.14
871	Hanworth	52	37	11	8.2	9.2	0.20	0.92	0.75	0.75	1.00	0.14	0.14
861	Isleham	67	13	20	6.2	8.5	0.20	0.76	0.75	0.75	0.97	0.11	0.11
642	Lakenheath	75	17	8	2.3	2.9	0.20	0.89	0.75	0.75	0.88	0.12	0.12
1025	Mendham	5	32	63	25	25	0.33	0.72	0.75	0.75	1.00	0.18	0.18
814	Newchurch	2	37	61	4.2	6	0.42	0.75	0.75	0.75	1.00	0.23	0.23
551	Newport	73	19	8	1.1	2.6	0.20	0.90	0.90	0.75	0.92	0.15	0.12
552	Ollerton	73	12	15	2.2	3.2	0.20	0.78	0.76	0.75	0.92	0.11	0.11
711	Prolleymoor	14	58	28	2.8	5	0.27	0.89	0.75	0.75	1.00	0.18	0.18
712	Ragdale	35	30	35	2.6	4.9	0.20	0.79	0.75	0.75	1.00	0.12	0.12
631	Redlodge	88	8	4	3.5	4.3	0.20	0.89	0.75	0.75	0.54	0.07	0.07
541	Sheringham	48	42	10	0.8	3.4	0.20	0.94	0.96	0.75	1.00	0.18	0.14
831	Sustead	54	33	13	2.1	2.6	0.20	0.91	0.76	0.75	1.00	0.14	0.14
813	Wallasea	9	41	50	4	5.5	0.28	0.79	0.75	0.75	1.00	0.16	0.16
541	Wick	59	27	14	1.7	3.5	0.20	0.88	0.78	0.75	0.99	0.14	0.13
572	Wickmere	29	42	29	0.9	2.5	0.20	0.85	0.94	0.75	1.00	0.16	0.13

USLE K Factor = fcsand x fcl-si x fhisand

fcl-si = (msilt/(mc + msilt))^3
forgc = (1 – (0.25 x orgc/(orgc + exp(3.72 – 2.95 x orgc))))

Where: msilt = % silt mc = % clay
ms = % sand orgC = % organic carbon

Table 5: E Calculations

				Erodibility (E) (max values) (t ha^{-1})					
Soil Association	Areas In SWAT (ha)	Ancillary Subgroups	Proportions	Winter Wheat/Winter Barley & Spring Barley	Potatoes/ Sugar Beet	Maize	Grass	fallow	Field Beans/ Peas
Wallasea 1	116.0625	813 Wallasea	75	2.46	3.15	3.35	0.01	4.92	0.59
		814 Newchurch	25	3.51	4.50	4.78	0.02	7.03	0.84
Newport 4	3787.6766	551 Newport	76	2.23	2.85	3.03	0.01	4.45	0.53
		631 Redlodge	24	1.08	1.38	1.47	0.01	2.16	0.26
Isleham 2	1463.0411	861 Isleham	31	1.66	2.12	2.25	0.01	3.31	0.40
		1024 Adventurers	29	0.93	1.19	1.27	0.01	1.87	0.22
		552 Ollerton	20	1.63	2.09	2.22	0.01	3.27	0.39
		821 Blackwood	20	1.88	2.41	2.56	0.01	3.76	0.45
Wick 3	17645.7177	541 Wick	61	2.07	2.65	2.82	0.01	4.14	0.50
		541 Sheringham	28	2.70	3.46	3.68	0.01	5.41	0.65
		551 Newport	11	2.23	2.85	3.03	0.01	4.45	0.53
Wick 2	31279.8211	541 Wick	38	2.07	2.65	2.82	0.01	4.14	0.50
		572 Wickmere	36	2.47	3.16	3.35	0.01	4.93	0.59
		541 Sheringham	16	2.70	3.46	3.68	0.01	5.41	0.65
		543 Aylsham	10	1.84	2.36	2.51	0.01	3.68	0.44
Hanworth	2038.9593	871 Hanworth	40	2.08	2.67	2.84	0.01	4.17	0.50
		831 Sustead	30	2.06	2.64	2.81	0.01	4.13	0.50
		1024 Adventurers	30	0.93	1.19	1.27	0.01	1.87	0.22
Gresham	1547.7441	711 Gresham	63	2.35	3.00	3.19	0.01	4.69	0.56
		711 Prolleymoor	21	2.68	3.42	3.64	0.02	5.35	0.64
		831 Sustead	16	2.06	2.64	2.81	0.01	4.13	0.50
Beccles 1	1039.7624	711 Beccles	65	1.96	2.51	2.66	0.01	3.92	0.47
		712 Ragdale	35	1.80	2.30	2.45	0.01	3.60	0.43
Altcar 2	621.5805	1022 Altcar	50	1.84	2.36	2.51	0.01	3.68	0.44
		1024 Adventurers	30	0.93	1.19	1.27	0.01	1.87	0.22
		1025 Mendham	20	2.65	3.39	3.60	0.02	5.30	0.64
Felthorpe	1674.0102	643 Felthorpe	40	2.46	3.15	3.35	0.01	4.93	0.59
		642 Lakenheath	27	1.79	2.29	2.43	0.01	3.57	0.43
		821 Blackwood	33	1.88	2.41	2.56	0.01	3.76	0.45

APPENDIX EIGHT
Example of CropWat Output

Potatoes

```
10/11/2004                                                 CropWat 4
Windows Ver 4.3
**********************************************************************
*********

                          Irrigation Scheduling Report

**********************************************************************
*********

* Crop Data:
------------
- Crop  # 1    : Potato
- Block #      : 1
- Planting date: 1/4

* Soil Data:
------------
- Soil description               : Light
- Initial soil moisture depletion: 0%


* Irrigation Scheduling Criteria:
---------------------------------
- Application Timing:
     Irrigate when 100% of readily soil moisture depletion occurs.
- Applications Depths:
     Refill to 100% of readily available soil moisture.
- Start of Scheduling: 1/4
```

Date	TAM (mm)	RAM (mm)	Total Rain (mm)	Efct. Rain (mm)	ETc (mm)	ETc/ETm (%)	SMD (mm)	Interv. (Days)	Net Irr. (mm)	Lost Irr. (mm)	User Adj. (mm)
1/4	30.0	7.5	7.6	0.0	0.9	100.0%	0.9				
6/4	32.7	8.3	7.3	4.6	1.0	100.0%	1.0				
11/4	35.5	9.2	7.0	5.0	1.0	100.0%	1.0				
16/4	38.2	10.1	6.7	5.4	1.1	100.0%	1.1				
21/4	40.9	11.0	6.4	5.8	1.2	100.0%	1.2				
26/4	43.6	11.9	6.2	6.2	1.4	100.0%	1.5				
1/5	46.4	12.9	5.9	5.9	1.8	100.0%	3.5				
6/5	49.1	13.8	5.7	5.7	2.2	100.0%	8.0				
9/5	50.7	14.4	0.0	0.0	2.5	100.0%	15.1	38	15.1	0.0	
11/5	51.8	14.8	5.5	2.6	2.7	100.0%	2.7				
16/5	54.5	15.9	5.3	5.3	3.2	100.0%	12.3				
18/5	55.6	16.3	0.0	0.0	3.4	100.0%	18.9	9	18.9	0.0	
21/5	57.3	16.9	5.2	5.2	3.7	100.0%	5.7				

```
25/5  59.5   17.8   0.0    0.0    4.2    100.0%  21.7    7    21.7
0.0
26/5  60.0   18.0   5.0    0.0    4.2    100.0%  4.2
30/5  60.0   18.0   0.0    0.0    4.4    100.0%  21.4    5    21.4
0.0
31/5  60.0   18.0   4.9    0.0    4.4    100.0%  4.4
4/6   60.0   18.0   0.0    0.0    4.5    100.0%  22.2    5    22.2
0.0
5/6   60.0   18.0   4.8    0.0    4.5    100.0%  4.5
8/6   60.0   18.0   0.0    0.0    4.6    100.0%  18.4    4    18.4
0.0
10/6  60.0   18.0   4.8    4.7    4.7    100.0%  4.7
13/6  60.0   18.0   0.0    0.0    4.8    100.0%  18.9    5    18.9
0.0
15/6  60.0   18.0   4.7    4.7    4.8    100.0%  4.9
18/6  60.0   18.0   0.0    0.0    4.9    100.0%  19.4    5    19.4
0.0
20/6  60.0   18.0   4.7    4.7    4.9    100.0%  5.1
23/6  60.0   18.0   0.0    0.0    5.0    100.0%  19.9    5    19.9
0.0
25/6  60.0   18.0   4.8    4.8    5.0    100.0%  5.2
28/6  60.0   18.0   0.0    0.0    5.0    100.0%  20.3    5    20.3
0.0
30/6  60.0   18.0   4.8    4.8    5.1    100.0%  5.3
3/7   60.0   18.0   0.0    0.0    5.1    100.0%  20.5    5    20.5
0.0
5/7   60.0   18.0   4.9    4.9    5.1    100.0%  5.3
8/7   60.0   18.0   0.0    0.0    5.1    100.0%  20.6    5    20.6
0.0
10/7  60.0   18.4   5.0    5.0    5.1    100.0%  5.1
13/7  60.0   19.6   0.0    0.0    4.9    100.0%  19.9    5    19.9
0.0
15/7  60.0   20.4   5.2    4.8    4.7    100.0%  4.7
19/7  60.0   22.0   0.0    0.0    4.5    100.0%  23.1    6    23.1
0.0
20/7  60.0   22.4   5.4    0.0    4.4    100.0%  4.4
25/7  60.0   24.4   5.6    5.6    4.1    100.0%  19.9
27/7  60.0   25.2   0.0    0.0    3.9    100.0%  27.9    8    27.9
0.0
30/7  60.0   26.4   5.8    5.8    3.7    100.0%  5.6
4/8   60.0   28.4   6.0    6.0    3.4    100.0%  17.1
---------------------------------------------------------------------
---------
Total               145.2  107.6  445.7 100.0%               308.3
0.0   0.0
---------------------------------------------------------------------
---------

* Yield Reduction:
------------------
- Estimated yield reduction in growth stage # 1 = 0.0%
- Estimated yield reduction in growth stage # 2 = 0.0%
- Estimated yield reduction in growth stage # 3 = 0.0%
- Estimated yield reduction in growth stage # 4 = 0.0%
                                                 --------
- Estimated total yield reduction                = 0.0%

* These estimates may be used as guidelines and not as actual figures.
---------------------------------------------------------------------
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* Legend:
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 TAM = Total   Available Moisture = (FC% - WP%)* Root Depth  [mm].
 RAM = Readily Available Moisture = TAM * P                  [mm].
 SMD = Soil Moisture Deficit                                 [mm].

* Notes:
--------
 Monthly ETo is distributed using polynomial curve fitting.
 Monthly Rainfall is distributed using polynomial curve fitting.
 To generate rainfall events, each 5 days of distributed rainfall are
  accumulated as one storm.
 Only NET irrigation requirements are given here. No any kind of
losses
  was taken into account in the calculations.
*********************************************************************
*********
C:\CROPWATW\REPORTS\PTS_L.TXT
```

APPENDIX NINE
EA Abstraction Data for SWAT

	Non-specified Use (m^3 day^{-1})						Agriculture (m^3 day^{-1})							
	Surface		Borehole		Well		Surface		Borehole		Well		Total (m^3 day^{-1})	
Subcatchment	EA	80%	EA	80%	EA	80%	EA	80%	EA	80%	EA	80%	EA	80%
1	0.00	0.00	59795.00	47836.00	4.56	3.65	573.00	458.40	13.60	10.88	0.00	0.00	60386.16	48308.93
2	0.00	0.00	5219.11	4175.29	0.00	0.00	0.00	0.00	0.00	0.00	20.42	16.34	5239.53	4191.62
3	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	282.54	226.03	0.00	0.00	282.54	226.03
4	754.00	603.20	99.62	79.70	0.00	0.00	0.00	0.00	1082.49	865.99	7.40	5.92	1943.51	1554.81
5	1501.00	1200.80	0.00	0.00	0.00	0.00	0.00	0.00	7.00	5.60	0.00	0.00	1508.00	1206.40
6	1000.00	800.00	60.00	48.00	10.00	8.00	0.00	0.00	2915.00	2332.00	18.14	14.51	4003.14	3202.51
7	2046.00	1636.80	243.60	194.88	0.00	0.00	0.00	0.00	31.80	25.44	31.80	25.44	2353.20	1882.56
8	523.00	418.40	11853.82	9483.06	1649.10	1319.28	910.00	728.00	6.80	5.44	8.80	7.04	14951.52	11961.22
9	1282.00	1025.60	1449.00	1159.20	1052.00	841.60	0.00	0.00	1899.50	1519.60	19.04	15.23	5701.54	4561.23
10	0.00	0.00	11585.00	9268.00	1.36	1.09	900.00	720.00	3370.82	2696.66	0.00	0.00	15857.18	12685.74
11	4090.00	3272.00	5471.35	4377.08	0.00	0.00	655.00	524.00	5974.00	4779.20	47.64	38.11	16237.99	12990.39
12	1945.00	1556.00	6.82	5.46	0.00	0.00	0.00	0.00	4073.50	3258.80	1123.00	898.40	7148.32	5718.66
13	3757.00	3005.60	1635.28	1308.22	288.64	230.91	818.00	654.40	2496.59	1997.27	2.96	2.37	8998.47	7198.78
14	0.00	0.00	2452.00	1961.60	82.00	65.60	1365.00	1092.00	898.60	718.88	0.00	0.00	4797.60	3838.08
15	0.00	0.00	34.00	27.20	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	34.00	27.20
16	2675.00	2140.00	13099.36	10479.49	1814.74	1451.79	0.00	0.00	2012.20	1609.76	1001.83	801.46	20603.13	16482.50
17	190500.00	152400.00	2795.00	2236.00	0.00	0.00	0.00	0.00	3706.90	2965.52	0.00	0.00	197001.90	157601.52
18	0.00	0.00	8310.00	6648.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	8310.00	6648.00
19	0.00	0.00	2.30	1.84	0.00	0.00	0.00	0.00	0.00	0.00	15.00	12.00	17.30	13.84
20	0.00	0.00	10.00	8.00	0.00	0.00	0.00	0.00	820.00	656.00	0.00	0.00	830.00	664.00
21	8572.00	6857.60	2751.30	2201.04	0.00	0.00	0.00	0.00	1745.60	1396.48	0.00	0.00	13068.90	10455.12
22	7336.73	5869.38	45.49	36.39	3.41	2.73	0.00	0.00	13598.30	10878.64	958.00	766.40	21941.93	17553.54
23	3881.00	3104.80	2360.00	1888.00	602.27	481.82	0.00	0.00	17468.25	13974.60	614.09	491.27	24925.61	19940.49
24	4296.00	3436.80	25174.00	20139.20	3445.12	2756.10		0.00	101.64	81.31	15.96	12.77	33032.72	26426.18
25	1090.00	872.00	1669.70	1335.76	1.10	0.88	0.00	0.00	56.50	45.20	7.70	6.16	2825.00	2260.00
26	2072.00	1657.60	1188.50	950.80	1273.00	1018.40	0.00	0.00	3738.82	2991.06	27.10	21.68	8299.42	6639.54
27	3180.00	2544.00	10502.64	8402.11	0.00	0.00	0.00	0.00	2021.55	1617.24	15.00	12.00	15719.19	12575.35
28	383.00	306.40	69252.36	55401.89	23.00	18.40	0.00	0.00	1049.00	839.20	0.00	0.00	70707.36	56565.89
29	1320.00	1056.00	3874.16	3099.33	2727.00	2181.60	0.00	0.00	1461.46	1169.17	5.00	4.00	9387.62	7510.10

APPENDIX TEN
EA Discharge Data for SWAT

EA discharge data for the River Ant

NUMBER	RECEIVING_	DWF	MAX_DAILY_	EASTING	NORTHING
AEENF1312	River Ant NT	2600.00	6618.00	635710	324280
PR4NF660X	Trib River Ant	0.00	3409.00	629900	325200
PR4NF1129X	Via a Weir Ormesby Broad	0.00	1500.00	647200	315300
AEENF12002	Hundred Stream River Ant	17.00	45.00	633890	327870
PRENF3708	Trib River Ant	0.00	16.00	634370	327360
PR4NF270	Trib River Ant	0.00	15.00	634100	329700
PR4NF1112X	Trib River Ant	0.00	10.00	634100	329700
PR4NF1560	Trib River Ant	0.00	10.00	631870	323270
PR4NF568	tributary River Ant	0.00	10.00	632700	327600
PRENF327	tributary River Ant	0.00	8.00	635430	332000
PR4NF1682	Trib River Ant	0.00	5.00	638200	324600
PRENF2562	Trib Dilhan Canal	0.00	5.00	630300	329250
PRETF8563	River Ant	0.00	5.00	634820	324830
PR4NF2084	Trib River Ant	0.00	4.00	625800	333900
PR4NF284	Trib River Ant	0.00	4.00	633020	324600
PRENF127	Trib Dilham Canal	0.00	4.00	632280	326420
PR4NF162	Limekiln Dyke River Ant	0.00	3.00	634420	321090
PRENF63	Trib River Ant	0.00	3.00	634390	320130
PRENF755	Trib River Ant	0.00	3.00	633060	327710
PR4NF1084	Trib River Ant	0.00	2.00	634820	327660
PR4NF1095X	Trib River Ant	0.00	2.00	635200	319100
PR4NF1871	Trib River Ant	0.00	2.00	632700	327500
PR4NF1952	Trib River Ant	0.00	2.00	628700	331900
PR4NF309	Limekiln Dyke Barton Broa	0.00	2.00	632300	321000
PR4NF426	Limekiln Dyke	0.00	2.00	634420	321090
PR4NF678X	Trib of River Ant	0.00	2.00	631100	329700
PR4NF886	Trib River Ant	0.00	2.00	627200	331600
PR4NF913	Trib River Ant	0.00	2.00	636400	326300
PRENF10334	tributary River Ant	0.00	2.00	636750	326250
PRENF2238	Trib North Walsham & Dilh	0.00	2.00	633400	327270
PRENF3426	Trib Dilham Canal	0.00	2.00	632950	326650
PRENF8736	tributary River Ant	0.00	2.00	629620	325880
PRENF8857	tributary River Ant	0.00	2.00	632610	327660
PRENF11640	TRIBUTARY OF RIVER ANT		2.00	634200	320180
PRENF11669	LIMEKILN DYKE		2.00	634370	321020
PR4NF1035	Trib River Ant	0.00	1.00	622300	332600
PR4NF1657	Unknown Trib River Ant	0.00	1.00	627110	333480
PR4NF1966	Trib River Ant	0.00	1.00	632600	327600
PR4NF1976	Trib River Ant	0.00	1.00	634300	321000
PR4NF1978	Trib River Ant	0.00	1.00	627300	333500
PRENF10135	tributary River Ant	0.00	1.00	631500	327400
PRENF10136	tributary River Ant	0.00	1.00	634550	324180
PRENF10219	tributary River Ant	0.00	1.00	637400	328800
PRENF10224	tributary River Ant	0.00	1.00	630000	328500
PRENF10379	North Walsham & Dilham Ca	0.00	1.00	630000	330650
PRENF10854	tributary River Ant	0.00	1.00	634090	321110
PRENF11408	tributary River Ant	0.00	1.00	634500	324220
PRENF152	Trib Dilham Canal	0.00	1.00	632690	327680
PRENF1534	Trib River Ant	0.00	1.00	631540	327330

PRENF155	Trib Dilham Canal	0.00	1.00	632690	327680
PRENF179	Trib River Ant	0.00	1.00	634280	320940
PRENF180	Trib River Ant	0.00	1.00	634280	320940
PRENF220	Trib River Ant	0.00	1.00	632620	327600
PRENF4089	Trib River Ant	0.00	1.00	634320	320180
PRENF8347	Trib River Ant	0.00	1.00	633920	324090
PRENF8468	Trib River Ant	0.00	1.00	632730	324800
PRENF8714	tributary River Ant	0.00	1.00	634110	321090
PRENF952	Trib River Ant	0.00	1.00	634800	330180
PRENF11750	North Walsham & Dilham Canal		1.00	633000	327700
PRENF11812	TRIBUTARY OF RIVER ANT		1.00	633200	325300
PRENF11813	TRIBUTARY OF RIVER ANT		1.00	633200	325301
PRENF13199	BARTON BROAD		1.00	635000	321310
AEENF1202	Foxes Beck River Ant	160.00	0.00	626540	334850
AW4NF868	River Ant NT	25.00	0.00	630300	325700
PR4NF751X	Trib River Ant	23.00	0.00	630000	320000
AW4NF637X	Tributary Rive Ant NT	15.00	0.00	633000	324500
AW4NF1082X	Ditch to River Ant NT	0.00	0.00	632800	327700
AWENF103	Tributary River Ant NT	0.00	0.00	634610	330770
PR4NF571	Trib River Ant	0.00	0.00	634900	324900

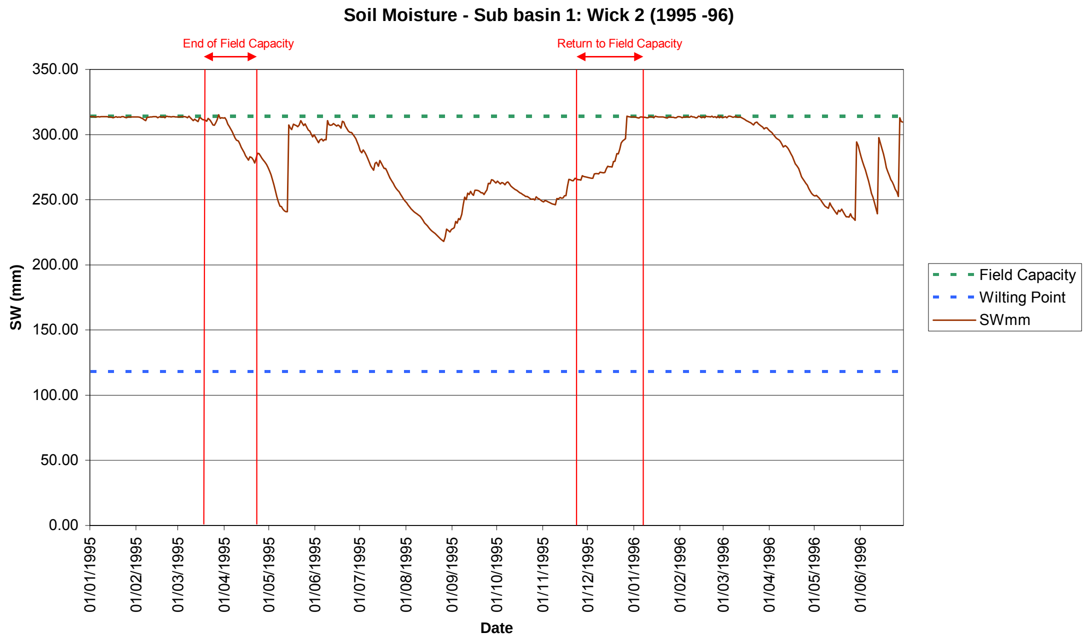
EA discharge data for the River Bure

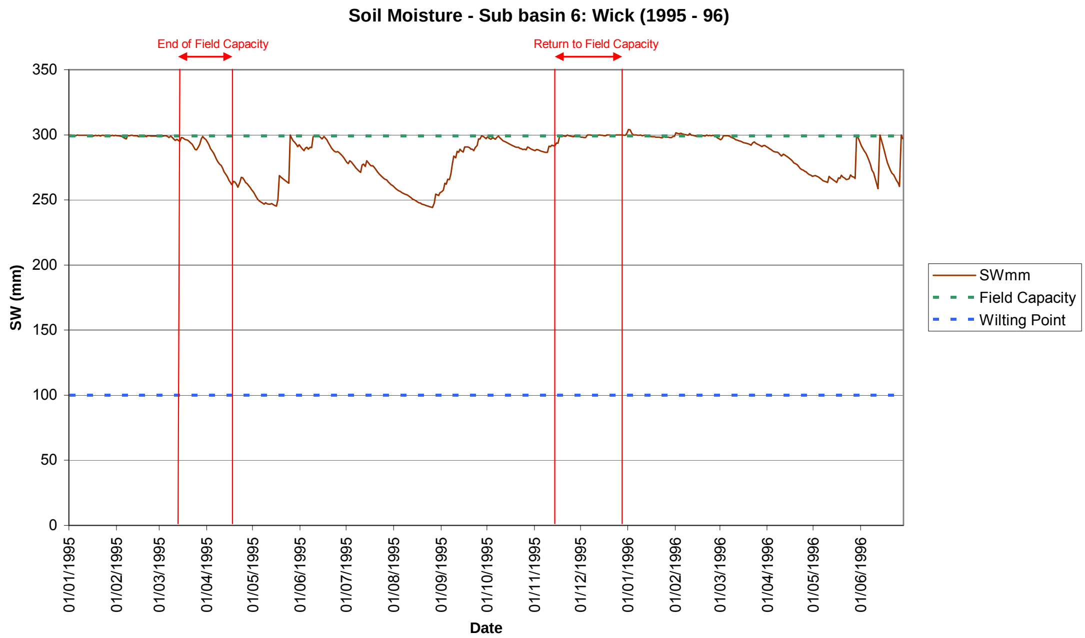
NUMBER	RECEIVING_	DWF	MAX_DAILY_	EASTING	NORTHING
PR4NF1656	River Bure	0.00	1000.00	614800	330600
AEENF12058	Scarrow Beck River Bure	291.00	755.00	618580	333590
PRENF10038	River Bure	0.00	500.00	624730	321840
CDENF10411	River Bure	0.00	323.00	624700	321900
PR4NF972	Trib River Bure	0.00	32.00	603100	331700
PR4TF527X	tidal River Bure	0.00	9.00	643900	309000
PR4TS527X	River Bure	0.00	9.00	643900	309000
PR4NF1075X	Trib River Bure	0.00	5.00	614500	330800
PR4NF789	Trib River Bure	0.00	5.00	626400	321000
PRENF10753	tributary River Bure	0.00	5.00	605930	326730
PR4NF1488	tributary River Bure	0.00	4.00	603600	331400
PR4NF863X	Trib Scarrow Beck	0.00	4.00	616700	333300
PRENF10059	tributary River Bure	0.00	4.00	619150	320690
PRENF10702	catchment of River Bure	0.00	4.00	634600	311500
PRENF10750	tributary River Bure	0.00	4.00	609710	333070
PRENF11403	tributary River Bure	0.00	4.00	640400	306600
PR4NF810X	River Bure	0.00	3.00	613900	333200
PRENF10388	tributary Scarrow Beck	0.00	3.00	619020	331180
PRENF8981	tributary River Bure	0.00	3.00	612820	334990
PR4NF1049X	Trib River Bure	0.00	2.00	630700	317400
PR4NF1334	Trib River Bure	0.00	2.00	609920	333720
PR4NF1338	Trib River Bure	0.00	2.00	644800	315300
PR4NF1479	Watercourse River Bure	0.00	2.00	643200	310500
PR4NF1667	Watercourse River Bure	0.00	2.00	612250	329600
PR4NF1799	Trib River Bure	0.00	2.00	614050	333330

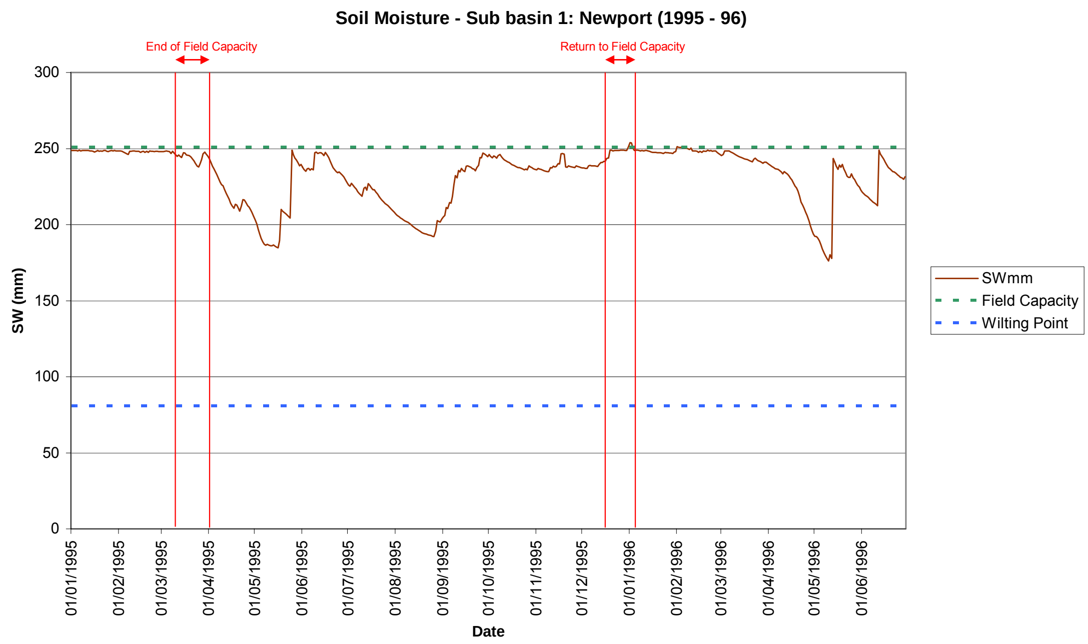
PR4NF2062	Trib River Bure	0.00	2.00	633700	312100
PR4NF262X	tributary River Bure	0.00	2.00	643200	310500
PR4NF608	Trib River Bure	0.00	2.00	622800	334600
PR4NF769	Stream River Bure	0.00	2.00	622600	329800
PRENF10245	tributary River Bure	0.00	2.00	615200	327500
PRENF10725	River Bure	0.00	2.00	614100	330400
PRENF10850	River Bure	0.00	2.00	633890	317650
PRENF11112	tributary Scarrow Beck	0.00	2.00	619300	334360
PRENF3469	River Bure	0.00	2.00	615410	329990
PRENF7834	Trib of River Bure	0.00	2.00	622530	329760
PRENF8846	River Bure	0.00	2.00	627600	319660
PRETF10087	River Bure	0.00	2.00	630750	317570
PRETF2495	River Bure	0.00	2.00	633720	317410
PRENF11833	TRIBTUARY OF RIVER BURE		2.00	618480	327100
PRENF13331	TRIBUTARY RIVER BURE		2.00	621350	326270
PR4NF1228	unknown trib River Bure	0.00	1.00	644800	315300
PR4NF1524	River Bure	0.00	1.00	610500	330700
PRENF10048	tributary River Bure	0.00	1.00	633860	317750
PRENF10080	tributary River Bure	0.00	1.00	618330	315750
PRENF10190	tributary River Bure	0.00	1.00	608250	331750
PRENF10914	tributary River Bure	0.00	1.00	609700	333200
PRENF11013	tributary Scarrow Beck	0.00	1.00	618110	334660
PRENF11109	tributary River Bure	0.00	1.00	639270	307580
PRENF11249	tributary River Bure	0.00	1.00	603860	329440
PRENF11250	tributary River Bure	0.00	1.00	603860	329440
PRENF11251	tributary River Bure	0.00	1.00	603860	329440
PRENF11252	tributary River Bure	0.00	1.00	603860	329440
PRENF11297	tributary River Bure	0.00	1.00	636300	312500
PRENF11382	tributary River Bure	0.00	1.00	643270	310600
PRENF11478	River Bure	0.00	1.00	634200	317250
PRENF130	Trib River Bure	0.00	1.00	604220	329450
PRENF68	River Bure	0.00	1.00	630900	317510
PRENF750	River Bure	0.00	1.00	607050	331780
PRENF8189	Trib River Bure	0.00	1.00	640900	312750
PRENF8426	Trib River Bure	0.00	1.00	614000	333320
PRENF8603	Trib River Bure	0.00	1.00	632720	316150
PRENF8896	tributary Tidal River Bur	0.00	1.00	627930	319350
PRENF8974	tributary River Bure	0.00	1.00	618270	339780
PRELF13351	TRIBUTARY RIVER BURE		1.00	613200	327500
PRENF11584	TRIBUTARY OF RIVER BURE		1.00	610730	329850
PRENF11783	TRIBUTARY OF RIVER BURE		1.00	621400	326200
PRENF11853	TRIBUTARY OF SCARROW BECK		1.00	615850	337000
PRENF11953	RIVER BURE		1.00	613990	333300
PRENF13239	RIVER BURE		1.00	630610	317510
AW4NF550X	R.Bure NT	818.00	0.00	620500	326700
AEETF1000	River Bure T	720.00	0.00	640900	309800
AW4TF303AX	Tributary of River Bure T	187.00	0.00	636900	313800
AW4NF795	River Bure NT	108.00	0.00	611800	329900
AEENF100	Stakebridge Beck River Bu	0.00	0.00	626490	325030

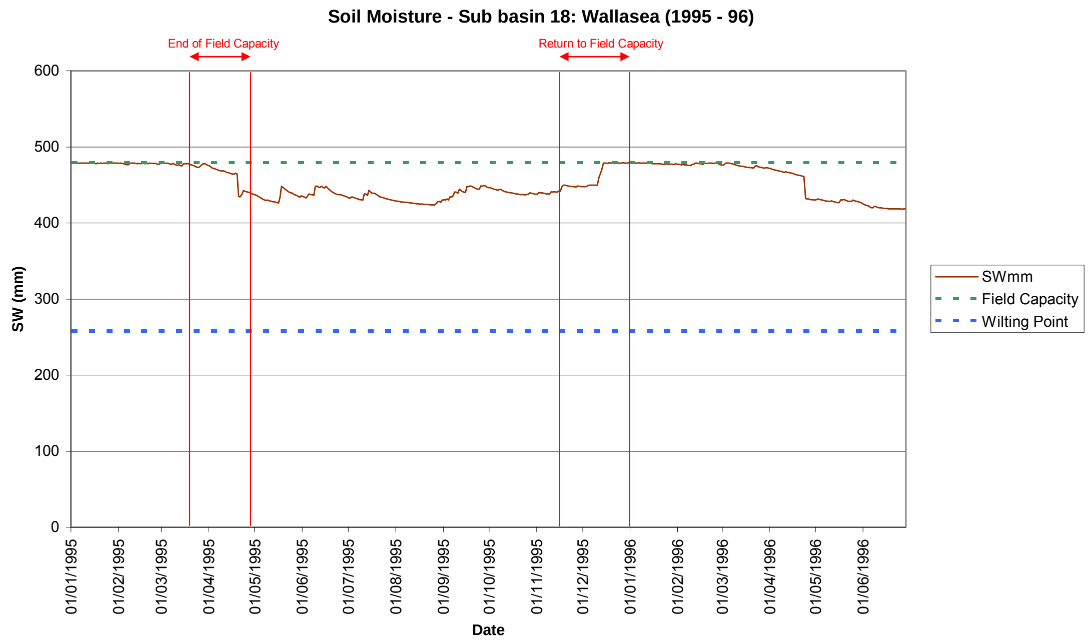
AW4NF1090 X	Dt to Trib of Kings Beck	0.00	0.00	624100	326700
PR4TF1523	River Bure	0.00	0.00	630400	318200

APPENDIX ELEVEN
SWAT Soil Moisture Outputs


Soil Moisture - Sub basin 1: Wick 2 (1995 -96)
End of Field Capacity
Return to Field Capacity
350.00
300.00
250.00
200.00
150.00
100.00
50.00
0.00
SW (mm)
01/01/1995
01/02/1995
01/03/1995
01/04/1995
01/05/1995
01/06/1995
01/07/1995
01/08/1995
01/09/1995
01/10/1995
01/11/1995
01/12/1995
01/01/1996
01/02/1996
01/03/1996
01/04/1996
01/05/1996
01/06/1996
Date
Field Capacity
Wilting Point
SWmm


Soil Moisture - Sub basin 6: Wick (1995 - 96)
End of Field Capacity
Return to Field Capacity
350
300
250
200
150
100
50
0
SW (mm)
01/01/1995
01/02/1995
01/03/1995
01/04/1995
01/05/1995
01/06/1995
01/07/1995
01/08/1995
01/09/1995
01/10/1995
01/11/1995
01/12/1995
01/01/1996
01/02/1996
01/03/1996
01/04/1996
01/05/1996
01/06/1996
Date
SWmm
Field Capacity
Wilting Point


Soil Moisture - Sub basin 1: Newport (1995 - 96)
End of Field Capacity
Return to Field Capacity
SW (mm)
300
250
200
150
100
50
0
01/01/1995
01/02/1995
01/03/1995
01/04/1995
01/05/1995
01/06/1995
01/07/1995
01/08/1995
01/09/1995
01/10/1995
01/11/1995
01/12/1995
01/01/1996
01/02/1996
01/03/1996
01/04/1996
01/05/1996
01/06/1996
Date
SWmm
Field Capacity
Wilting Point


Soil Moisture - Sub basin 18: Wallasea (1995 - 96)
End of Field Capacity
Return to Field Capacity
SW (mm)
600
500
400
300
200
100
0
01/01/1995
01/02/1995
01/03/1995
01/04/1995
01/05/1995
01/06/1995
01/07/1995
01/08/1995
01/09/1995
01/10/1995
01/11/1995
01/12/1995
01/01/1996
01/02/1996
01/03/1996
01/04/1996
01/05/1996
01/06/1996
Date
SWmm
Field Capacity
Wilting Point