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Investigation of the Impact of Land Use Management Scenarios on Diffuse Source Nutrients in the River Axe Catchment

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ABSTRACT

The Water Framework Directive (WFD) (Directive 2000/60/EC) requires new ecological standards for rivers, lakes and coastal waters by 2015. In the United Kingdom the English Catchment Sensitive Farming Initiative has identified 40 catchments which are at risk of failing the European Commission WFD targets for good ecological status of water bodies because of a range of issues. The river Axe catchment situated in south-west part of the UK, with a mixture of diffuse and point sources of pollution, is one of these priority sites, as intensive dairy farming and cultivation of high risk crops (maize), cause problems with enhanced suspended sediment, nitrate and phosphorus levels in the river. For the purpose of this research the Soil and Water Assessment Tool (SWAT-2005) and ArcView GIS 3.2 interface AVSWAT-X were used. SWAT has been found to be a useful tool in numerous EU and UK studies addressing the objectives of the EU WFD, which requires identification of pollution sources, their influences and solutions for the studied catchment. The base scenario was based on field observation and interviews with the Environment Agency and farmers; it was run with and without point sources. Model was performing well; achieving hydrological calibration for Nash-Sutcliffe efficiency coefficient (E_{NS}) of 0.62. Three different scenarios, field buffer strips (FBS), extensive land use management (ELUM) and sheep land use management (SLUM), were used to evaluate the ability of SWAT to represent the proposed mitigation methods and assess the effectiveness of the measures in reducing nutrient loads in the Axe. SWAT was found to be able to represent the proposed mitigation methods which were included in these three management scenarios. Management scenarios reduced the average annual loads at the main catchment outlet by 21.21% (FBS), 37.32% (ELUM) and 45.02% (SLUM), for total nitrogen; 47.72% (FBS), 60.58% (ELUM) and 62.41% (SLUM) for total phosphorus; and 3.49% (FBS), 7.34% (ELUM) and 5.58% (SLUM) for annual average sediment yield. To deliver the necessary water quality improvements for the river Axe both diffuse and point sources of pollution will have to be addressed as results show that reduction only in one type of source does not achieve the Environment Agency's Environmental Quality Standard target values which were set to meet the objectives of the EU WFD.

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TABLE OF CONTENTS

ABSTRACT I
ACKNOWLEDGEMENTS II
TABLE OF CONTENTS III
LIST OF TABLES V
LIST OF FIGURES VIII
LIST OF APPENDICES XII
LIST OF SYMBOLS XIV
ABBREVIATIONS AND ACRONYMS XVIII
1 INTRODUCTION 1
1.1 BACKGROUND 1
1.2 AIM AND OBJECTIVES 3
2 LITERATURE REVIEW 4
2.1 RIVER ECOSYSTEM 4
2.2 DIFFUSE SOURCE NUTRIENTS 7
2.2.1 Nitrogen (N) 8
2.2.2 Phosphorus (P) 10
2.3 DIFFUSE NUTRIENTS TRANSFER 13
2.3.1 Surface runoff 13
2.3.2 Infiltration and lateral subsurface flow 14
2.3.3 Groundwater leaching 15
2.4 CATCHMENT LAND MANAGEMENT 16
2.4.1 Best management practices 17
2.4.2 European Union Common Agricultural Policy 22
2.4.3 Land Management Scenarios 23
2.5 MODELLING OF DIFFUSE POLLUTANTS 25
2.5.1 Modelling Approaches 25
2.5.2 Soil and Water Assessment Tool 27
2.5.3 SWAT Applications 29
3 METHODOLOGY 31
3.1 STUDY AREA AND AVAILABLE DATA 31
3.1.1 Topography and Geography 31
3.1.2 Geology and Soils 33
3.1.3 Hydrology and Hydrogeology 35
3.1.4 Climate 40
3.1.5 Land Use 44
3.1.6 Livestock 46
3.1.7 Water Quality and Sources of Pollution 47
3.1.8 Designations 52
3.2 DATA COLLECTION, ANALYSIS AND MODEL BUILD 53
3.2.1 Model Requirements 53
3.2.2 Sub-catchment Delineation 56
3.2.3 Climate Data 58
3.2.4 Land use 61
3.2.5 Soils 62
3.2.6 Management Files 65
3.2.7 Abstraction Data 71

3.2.8 Discharge Data ... 73
4 CALIBRATION AND VALIDATION ... 75
4.1 MODELLING STRATEGY ... 76
4.2 MODEL PERFORMANCE STATISTICAL CRITERIA ... 77
4.3 CALIBRATION ... 80
4.3.1 Hydrological Calibration ... 80
4.3.1.1 Annual Flow Summaries ... 83
4.3.1.2 Daily Time Flow Series ... 85
4.3.2 Model Performance Indicators ... 88
4.3.3 Sediment Calibration ... 93
4.3.4 Nutrient Calibration ... 94
4.3.4.1 Nitrate Calibration ... 96
4.3.4.2 Phosphorus Calibration ... 104
4.4 VALIDATION ... 105
5 LAND MANAGEMENT SCENARIOS ... 112
5.1 SCENARIO 1 – FIELD BUFFER STRIPS ... 113
5.2 SCENARIO 2 – EXTENSIVE LAND USE MANAGEMENT ... 116
5.3 SCENARIO 3 – SHEEP LAND USE MANAGEMENT ... 118
6 RESULTS AND DISCUSSION ... 120
6.1 BASELINE SCENARIO ... 121
6.1.1 Sub-catchment contribution ... 121
6.1.2 River loads ... 122
6.2 LAND USE MANAGEMENT SCENARIOS ... 123
6.2.1 River Flow ... 123
6.2.2 Sediment ... 127
6.2.3 Nitrogen ... 131
6.2.4 Phosphorus ... 134
6.2.5 Scenario Evaluation and Recommendations ... 139
7 CONCLUSIONS ... 144
REFERENCES ... 150
APPENDICES ... 168

LIST OF TABLES

Table 2.1: Sources and sinks for the nitrogen budget (Adapted from: Campbell et al., 2004; Connell, 2005, Koo, 2006). 9
Table 2.2: Phosphorus sources and environmental problems. (Source: Campbell et al., 2004) 11
Table 2.3: BMP-s measures that offer general benefits for nitrogen, phosphorus and suspended solids (Source: Campbell et al., 2004) 18
Table 2.4: Typical phosphorous uptake of common crops (Source: Campbell and Edwards, 2000) 19
Table 2.5: Foresight Programme linked with the future Agricultural Policy Scenarios (Source: Morris, 2003) 24
Table 3.1: Environment Agency EQS nitrate nitrogen and phosphorus classification scheme (MacDonald, 2006; Environment Agency, 2006-b). 48
Table 3.2: Relationship between threshold area, number of HRU-s and percentages of different land use types based on 25 sub-catchments of the river Axe catchment study area. 55
Table 3.3: Relationships between CORINE and SWAT land use classes used in model. 61
Table 3.4: Percent (%) of arable and total study area covered by crop rotations in the model. 66
Table 3.5: Percent (%) and area (ha) of crops used in SWAT arable rotations. 67
Table 3.6: Percent (%) of pasture and total study area covered by pasture rotations in the model. 69
Table 3.7: Adopted average grazing parameters included in SWAT management files (Adapted from: Frame, 1992). 70
Table 3.8: Grazing parameters for dairy cow and sheep per HRU included in SWAT grazing management files. 70
Table 4.1: Calibration and validation data available for the study area. 77
Table 4.2: Model calibration hydrology criteria by Montana Department of Environment Quality (2005) 80
Table 4.3: Initial and final values of the calibrated hydrological parameters in SWAT model. 82
Table 4.4: Annual summary statistics for the calibration period (1988 – 1997) at Whitford; sub-catchment 10 outlet. 83
Table 4.5: Measured and simulated base flow results for the calibration period (1988 – 1997) as calculated by ABFS. 84
Table 4.6: Measured and simulated groundwater flow contribution to total stream flow for the calibration period (1988-1997) at Whitford sub-catchment 10 outlet, using ABFS method. 84
Table 4.7: Daily time step stream flow model performance statistic for the Whitford gauging station in the river Axe catchment for the calibration period (1988 – 1997). 85
Table 4.8: Acceptable model calibration hydrology criteria for the Whitford gauging station during calibration period (1988-1997). 87
Table 4.9: Initial and final values of SWAT model parameters for the calibration of nutrients. 95

Table 4.10: Daily time step stream flow model performance statistics for the Whitford gauging station in the river Axe catchment for the validation period (1998 – 2005). 106
Table 4.11: Comparison of measured and modelled validation results for mean annual concentration of nitrate nitrogen and ortho-phosphorus for four catchment sections. 111
Table 5.1: SWAT parameters used for baseline and new scenarios. 113
Table 5.2: SWAT parameters for modelling riparian zones and buffer strips. ... 114
Table 5.3: Grazing parameters for dairy cow and sheep per HRU, included in SWAT management files. 117
Table 6.1: Baseline average annual contribution of sediment yield in surface runoff transported into the reach for the simulation period and variability among catchment section sub-catchments. 121
Table 6.2: Baseline scenario average annual total nitrogen yield in surface runoff transported into the reach for the simulation period and variability among catchment section sub-catchments. 122
Table 6.3: Baseline average annual contribution of total phosphorus yield in surface runoff transported into the reach for the simulation period and variability among catchment section sub-catchments. 122
Table 6.4: The baseline scenario average annual sediment load, total nutrient loads and river flow at the outlets of four catchment sections for the simulation period. 123
Table 6.5: Impacts of the scenarios on average annual water yield and contributions to stream flow and actual evapotranspiration from the average sub-catchment in *.sub output file for the simulation period. 124
Table 6.6: Impacts of the scenarios at the four catchment section outlets, percentage of change in average annual river flow from baseline scenario. 124
Table 6.7: Impacts of the scenarios for the four catchment sections, as percentage change in average annual sediment yields in runoff from the baseline scenario. .. 127
Table 6.8: Impacts of the scenarios at the four catchment section outlets, percentage change in average annual river loads of sediment from the baseline scenario. 129
Table 6.9: Impacts of the scenarios for the four catchment sections, as percentage change in average annual total nitrogen yields in runoff from the baseline scenario. 132
Table 6.10: Impacts of the scenarios at the four catchment section outlets, percentage of change in average annual total nitrogen river loads from baseline scenario. 132
Table 6.11: Impacts of the scenarios for the four catchment sections, as percentage change from the baseline scenario in average annual total phosphorus yield transported by surface runoff. 134
Table 6.12: Impacts of the scenarios at the four catchment section outlets, as percentage change in average annual total phosphorus river loads from the baseline scenario. 135
Table 6.13: Impacts of the land use management scenarios on average annual phosphorus yields transported in surface runoff into the river for the river Axe catchment. 136

Table 6.14: Impacts of the land use management scenarios on average annual mineral phosphorus load and concentration in the river Axe at main catchment outlet and change as percent of base scenario load. 137
Table 6.15: Comparison of baseline and land use management scenario impacts on mean annual sediment, mineral phosphorus and nitrate nitrogen concentrations (mg l^{-1}) for the study period compared with EQS target values. 141

LIST OF FIGURES

Figure 2.1: The nitrogen cycle (Source: Neitsch et al., 2005). 9
Figure 2.2: The phosphorus cycle (Source: Neitsch et al., 2005). 12
Figure 3.1: The river Axe catchment area showing major tributaries and urban areas (Source: Ash et al., 2005) 31
Figure 3.2: View from the eastern part of the river Axe catchment towards the west (Blackdown Hills). 32
Figure 3.3: View on the River Axe estuary and coastal chalk plateaux. 33
Figure 3.4: Geology of eastern Devon and the river Axe catchment (Duff and Smith, 1992) 34
Figure 3.5: Soil map of the river Axe catchment (SSLRC, 1983). 35
Figure 3.6: River network of the Axe catchment. 36
Figure 3.7: The River Axe daily mean flow at Whitford gauging station from January 1986 to December 2005. 37
Figure 3.8: The River Axe mean monthly flow at Whitford gauging station from January 1986 to December 2005. 37
Figure 3.9: The River Axe yearly mean daily flow at Whitford gauging station for the period from January 1986 to December 2005. 38
Figure 3.10: Hydrogeological map of the river Axe catchment (IGS, 1977). 39
Figure 3.11: Mean annual precipitation for nine rainfall gauge stations in the study area for the period from 1986 to 2005. 40
Figure 3.12: Total annual precipitation and 20 years annual average for Cricket Wildlife Park gauge station for the period of 1986 to 2005 (BADC, 2007) 41
Figure 3.13: Twenty years monthly average precipitation for Cricket Wildlife Park gauge station for the period from 1986 to 2005 (BADC, 2007) 41
Figure 3.14: Monthly average (maximum, minimum, and mean) temperature (°C) for Hemyock for the period 1986 to 2005. 42
Figure 3.15: Monthly average (maximum, minimum, and mean) temperature (°C) for Sidmouth for the period 1986 to 2005. 42
Figure 3.16: Maximum, minimum, and mean temperature (°C) for Hemyock for the period from 1986 to 2005. 43
Figure 3.17: Maximum, minimum, and mean temperature (°C) for Sidmouth for the period from 1986 to 2005. 43
Figure 3.18: Land use in the river Axe catchment (CLC, 2000) 44
Figure 3.19: Area (ha) covered by different crops between years 1969 and 2004 (EDL, 2006) 45
Figure 3.20: Fraction of individual crops in study area arable land for year 2000 (EDL, 2006) 45
Figure 3.21: Number of dairy cows in the River Axe catchment between 1993 and 2004 (EDL, 2006). 46
Figure 3.22: Number of sheep in the River Axe catchment between 1969 and 2004 (EDL, 2006) 47
Figure 3.23: Water quality grade for nitrate nitrogen (NO_3-N) at the River Axe tidal limit (Axe Bridge). 48
Figure 3.24: Measured concentration of nitrate nitrogen (NO_3-N) at Whitford gauging station for the period from 1986 to 2006. 49

Figure 3.25: Water quality grade for ortho-phosphorus at the River Axe tidal limit (Axe Bridge). 50
Figure 3.26: Comparison of 'current' situation and 'diffuse sources only' SIMCAT predictions for phosphorus levels in the River Axe. Observed values (+) have been retained in both scenarios for ease of comparison. The red dotted line denotes guideline standard. (Source: Ash et al., 2005) 51
Figure 3.27: Measured concentration of ortho-phosphorus (PO_4^{3-}) at Whitford gauging station for the period from 1986 to 2006. 51
Figure 3.28: Digital Elevation model (DEM) – in metres above sea level. 56
Figure 3.29: The river Axe catchment delineation into 25 sub-catchments and 4 catchment sections (River Yarty, River Coly, Upper Axe, Lower Axe). 58
Figure 3.30: Weather gauging stations used in SWAT modelling of the study area. 59
Figure 3.31: Modified land use map used in SWAT modelling of the study area. .. 62
Figure 3.32: Comparison of actual and modelled areas of crops in the study area. 67
Figure 3.33: Locations of water abstraction points in the river Axe catchment. ... 72
Figure 3.34: Locations of waste water discharge points in the river Axe catchment. 74
Figure 4.1: Base flow separation for measured data at Whitford using the ABFS method for the calibration period (1988 – 1997). 81
Figure 4.2: Detailed view of base flow separation at Whitford using the ABFS method for the calibration years 1994 and 1995. 82
Figure 4.3: Comparison of measured and simulated base flows (Pass 1) at Whitford for the calibration period (1988 – 1997). 84
Figure 4.4: Comparison between simulated and measured flow at Whitford for the calibration period (1988-1997). 86
Figure 4.5: Relationship between measured and simulated daily stream flow at Whitford for the calibration period (1988-1997) and corresponding coefficient of determination (R^2) 86
Figure 4.6: Comparison of cumulative distribution of simulated and measured flow on logarithmic scale for the calibration period (1988-1997) at Whitford gauging station. 87
Figure 4.7: Simulated evapotranspiration (Penman-Monteith method, data from Sidmouth) and measured temperature for the calibration period (1988-1997). 89
Figure 4.8: Comparison of simulated soil water content (mm) and measured precipitation (mm) for HRU 57 (Whimple 3, PSG4) during calibration period (1988-1997). 90
Figure 4.9: Comparison of published soil water content parameters (Findley et al., 1984; Smith, 1976) with simulated soil water content for HRU 57 (Whimple 3, PSG4) for calibration year 1990. 91
Figure 4.10: Modelled plant growth for winter wheat – ROT1 (Sub-catchment 1, HRU 5). 92
Figure 4.11: Comparison between SWAT simulated sediments (mg l^{-1}) and reported mean sediment concentration (mg l^{-1}) by EA (2004-a) for the year 2002. 94
Figure 4.12: Comparison between simulated daily nitrate nitrogen (NO_3-N) concentrations (SWAT) with measured nitrate nitrogen (NO_3-N) concentrations (EA) at Whitford for the calibration period (1988 – 1997). 97

Figure 4.13: Comparison between simulated daily nitrate nitrogen loads (kg N day^{-1}) (SWAT) with measured nitrate nitrogen loads (kg N day^{-1}) (EA) at Whitford for the calibration period (1988 – 1997). 98
Figure 4.14: Comparison of cumulative distribution of simulated (SWAT) and measured (EA) nitrate nitrogen (NO_3-N) concentrations at Whitford for the calibration period (1988 – 1997). 99
Figure 4.15: Comparison of cumulative distribution of simulated (SWAT) and measured (EA) nitrate nitrogen (NO_3-N) loads at Whitford for the calibration period (1988 – 1997). 99
Figure 4.16: Comparison of HRU output for nitrate loads in surface, lateral and groundwater flow and leaching (kg N ha^{-1}) for the calibration years from 1988 to 1992 in HRU 57. 100
Figure 4.17: Comparison of HRU 57 output for nitrate nitrogen (NO_3-N) in surface, lateral and groundwater flow and leaching (kg N ha^{-1}) for bad and good agreement year. 101
Figure 4.18: Comparison of measured and simulated nitrate nitrogen concentrations (mg NO_3-N l^{-1}) at the Whitford sampling point for bad and good agreement year. 101
Figure 4.19: River flow (m^3 s^{-1}) and load (kg N) of nitrate transported with water out of reach during time step for good and bad agreement year. 102
Figure 4.20: River flow (m^3 s^{-1}) and load of nitrate nitrogen transported with water out of reach during time step (kg) for calibration period and soil water content of the representative HRU 57. 102
Figure 4.21: Comparison between simulated daily mineral phosphorus (MINP) concentrations (SWAT) with measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) at Whitford for the calibration period (1988 – 1997). 104
Figure 4.22: Comparison of cumulative distribution of simulated mineral phosphorus (SWAT) concentrations and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) at Whitford for the calibration period (1988 – 1997). 105
Figure 4.23: Comparison between simulated and measured daily flow values (m^3 s^{-1}) for the Whitford gauging station for the validation period (1998 – 2005). 106
Figure 4.24: Comparison between daily simulated (SWAT) and measured (EA) concentrations of NO_3-N for the gauging station at Whitford during the validation period (1998 – 2005). 107
Figure 4.25: Comparison between simulated daily nitrate nitrogen (NO_3-N) loads (SWAT) with measured nitrate nitrogen (NO_3-N) loads (EA) at Whitford for the validation period (1998 – 2005) 108
Figure 4.26: Comparison of cumulative distribution of simulated (SWAT) and measured (EA) nitrate nitrogen (NO_3-N) concentrations at Whitford for the validation period (1998 – 2005). 108
Figure 4.27: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the gauging station at Whitford during the validation period (1998 – 2005) without phosphorus stripping introduction. 109
Figure 4.28: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the gauging station at Whitford (1998 – 2005) with phosphorus stripping introduction. 110

Figure 4.29: Comparison of cumulative distribution of simulated mineral phosphorus (SWAT) concentrations (with and without phosphorus stripping introduction) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) at Whitford for the validation period (1998 – 2005. 111
Figure 6.1: Average annual daily river flow change for the scenarios at the outlet of the four catchment sections as percent (%) of baseline scenario for the simulation period. 126
Figure 6.2: Comparison of average monthly daily river flow (m^3 s^{-1}) between baseline and management scenarios for the simulation period at the river Axe main catchment outlet. 126
Figure 6.3: Comparison of average annual sediment yield (t ha^{-1} $year^{-1}$) transported from the HRU into the main channel for the simulation period, between baseline and land use management scenarios. 128
Figure 6.4 Change in average annual load of sediment at the outlet of the four catchment sections as percent of baseline scenario for the simulation period. 129
Figure 6.5: Comparison of average monthly sediment concentrations (mg l^{-1}) between baseline and management scenarios for the simulation period at the Whitford gauging station – sub-catchment 10 outlet. 131
Figure 6.6: Change in average annual load of total nitrogen at the outlet of the four catchment sections as percent of baseline scenario for the simulation period. 133
Figure 6.7: Comparison of average monthly nitrate nitrogen loads (kg NO_3-N $month^{-1}$) at the river Axe main catchment outlet between baseline and management scenarios for the simulation period. 133
Figure 6.8: Change in load of total phosphorus at the outlet of the sub-catchments as percent of baseline scenario for the simulation period. 136
Figure 6.9: Comparison of average monthly mineral phosphorus loads (kg $month^{-1}$) in the river Axe at the main catchment outlet for baseline and land use management scenarios for the simulation period. 138
Figure 6.10: Comparison of average monthly mineral phosphorus concentrations (mg l^{-1}) at the river Axe main catchment outlet for baseline and management scenarios for the simulation period. 138

LIST OF APPENDICES

Appendix A1: Soil map of the river Axe catchment (SSLRC, 1983). 169
Appendix B1: Precipitation data for nine gauging stations used in the SWAT model (BADC, 2007). 170
Appendix C1: Data obtained for SWAT model build. 171
Appendix D1: General soil data in SWAT *.sol file. 172
Appendix D2: Layer 1 soil data in SWAT *.sol file. 173
Appendix D3: Layer 2 soil data in SWAT *.sol file. 174
Appendix D4: Layer 3 soil data in SWAT *.sol file. 175
Appendix D5: Layer 4 soil data in SWAT *.sol file. 176
Appendix D6: Layer 5 soil data in SWAT *.sol file. 177
Appendix E1: ROT1 (ww/wb/rg/rg/ww) crop management dates. 178
Appendix E2: ROT2 (m) crop management dates. 179
Appendix E3: ROT3 (ww/ww/sa/rg/rg/) crop management dates. 180
Appendix E4: ROT4 (rg/m/wb) crop management dates. 181
Appendix E5: ROT5 (rg/rg/rg/rg/rg) management dates. 182
Appendix E6: PA11 and PA12 sheep grazing management dates. 183
Appendix E7: PAS1 and PAS2 dairy cow grazing management dates. 183
Appendix E8: PSG4 grass management dates. 184
Appendix E9: ROTE (rc/rc/ww/wb) crop management dates. 184
Appendix E10: RTE5 (rg/rg/rg/rc/rc) management dates. 185
Appendix E11: PG4E extensive grass management dates. 186
Appendix E12: P11S and P12S extensive and sheep scenario grazing management dates. 186
Appendix E13: PG1S sheep scenario grass management dates. 187
Appendix E14: PG2S sheep scenario grass management dates. 187
Appendix F1: Distribution of crop rotations across arable (AGRC) HRU-s. 188
Appendix F2: Distribution of grass rotations across pasture (PAUK) HRU-s. 188
Appendix F3: Distribution of crop rotations across arable (AGRC) HRU-s for extensive scenarios 2. 189
Appendix F4: Distribution of rotations across arable (AGRC) HRU-s for sheep scenario 3. 189
Appendix G1: Fertiliser information in SWAT fert.dat file 190
Appendix H1: Water Abstraction data input in SWAT *.wus file 191
Appendix I1: Point source discharge input data used in SWAT without phosphorus stripping introduction. 192
Appendix I2: Point source discharge input data used in SWAT with phosphorus stripping introduction. 193
Appendix I3: Point source discharge data used for recalculation of SWAT input data. 194
Appendix J1: Plant growth parameters in SWAT database used in the model. 195
Appendix J2: Plant growth parameters in SWAT database used in the model. 196
Appendix K1: Modelled plant growth for permanent grass in three cut grass silage system - PSG4 (Sub-catchment 1, HRU 1). 197
Appendix K2: Modelled plant growth for winter barley – ROT4 (Sub-catchment 1, HRU 7). 197

Appendix K3: Modelled plant growth for silage maize – ROT2 (Sub-catchment 1, HRU 4). 198
Appendix K4: Modelled plant growth for rye grass – ROT5 (Sub-catchment 5, HRU 24). 198
Appendix L1: Comparison between daily simulated (SWAT) and measured (EA) concentrations of nitrate nitrogen (NO_3-N) for the Slymlakes sampling point during the validation period (1998 – 2005). 199
Appendix L2: Comparison between daily simulated (SWAT) and measured (EA) concentrations of nitrate nitrogen (NO_3-N) for the Newenham Abbey sampling point during the validation period (1998 – 2005). 199
Appendix L3: Comparison between daily simulated (SWAT) and measured (EA) concentrations of nitrate nitrogen (NO_3-N) for the Colyford sampling point during the validation period (1998 – 2005). 200
Appendix L4: Comparison between daily simulated (SWAT) and measured (EA) concentrations of nitrate nitrogen (NO_3-N) for the Axe Bridge sampling point during the validation period (1998 – 2005). 200
Appendix L5: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the Slymlakes sampling point during the validation period (1998 – 2005). 201
Appendix L6: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the Newenham Abbey sampling point during the validation period (1998 – 2005). 201
Appendix L7: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the Colyford sampling point during the validation period (1998 – 2005). 202
Appendix L8: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the Axe Bridge sampling point during the validation period (1998 – 2005). 202
Appendix M1: Comparison of average annual total nitrogen inputs (kg ha^{-1} $year^{-1}$) at HRU level averaged to sub-catchment level for the baseline and land use management scenarios. 203
Appendix M2: Comparison of average annual total nitrogen outputs (kg ha^{-1} $year^{-1}$) at HRU level averaged to sub-catchment level for the baseline and land use management scenarios. 204
Appendix N1: Comparison of average annual total phosphorus inputs (kg ha^{-1} $year^{-1}$) at HRU level averaged to sub-catchment level for the baseline and land use management scenarios. 205
Appendix N2: Comparison of average annual total phosphorus outputs (kg ha^{-1} $year^{-1}$) at HRU level averaged to sub-catchment level for the baseline and land use management scenarios. 206

LIST OF SYMBOLS

%	percent
<	less than
>	greater than
√	square root
°C	degrees Celsius
Al	aluminium
ALPHA_BF	alpha base flow factor
a_s	Angstrom regression constant = 0.25
a_s+b_s	fraction of extraterrestrial radiation reaching the earth on clear days (n = N)
AWC	available water capacity
BIOM	plant biomass (kg ha^{-1})
BIO_MIN	biomass below which grazing is not allowed
BIOMIX	bio-mixing efficiency
b_s	Angstrom regression constant = 0.50
cm	centimetres
C	carbon
Ca	calcium
CN2	curve number
C:N	carbon - nitrogen relationship
CO_2	carbon dioxide
c_p	specific heat at constant pressure (MJ kg^{-1} $°C^{-1}$)
E	depth rate evaporation (mm d^{-1})
e	accepted error measured as a proportion of the standard deviation
e_d	vapour pressure (kPa)
E_{NS}	Nash and Sutcliffe simulation efficiency
E_o	potential evapotranspiration (mm d^{-1})
ESCO	soil evaporation compensation factor
es_{db}	saturation vapour pressure (kPa)
es_{wb}	saturation vapour pressure (kPa)
ET	evapotranspiration
et al.	and others
$e°_z$	saturation vapor pressure of air at height z (kPa)
e_z	water vapor pressure of air at height z (kPa)
Fe	iron
$f_{cl\text{-}si}$	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_SURFACE	fertiliser application rates
FILTERW	width of edge-of-field filter strip
G	heat flux density to the ground (MJ m^{-2} d^{-1})
GRZ_DAYS	number of grazing days

GO_DELAY	groundwater delay time for base flow to occur
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
HI_OVR	harvest index override ((kg ha^{-1})(ka ha^{-1})$^{-1}$)
HLIFE_NGW	half-life of nitrate in the shallow aquifer (days)
H_{net}	net radiation (MJ m^{-2} d^{-1})
K	hydraulic conductivity (mm h^{-1})
kg	kilogram
kg ha^{-1}	kilograms per hectare
km	kilometre
km^2	kilometre squared
kPa	kilo Pascal's
K_{SAT}	saturated hydraulic conductivity (mm h^{-1})
K_{USLE}	soil erodibility factor
kWh	kilowatts per hour
l s^{-1}	litres per second
m	meter
MANURE_KG	dry weight of manure deposited daily
m_c	soil clay content (%)
mg	milligrams
mg kg^{-1}	milligrams per kilogram
mg l^{-1}	milligrams per litre
MINP_OUT	mineral phosphorus transported with water out of reach during time step (kg P)
mm	millimetres
mm hr^{-1}	millimetres per hour
m yr^{-1}	metres per year
m s^{-1}	metres per second
m^3 s^{-1}	metres cubed per second
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 (%)
n	actual duration of sunshine (hour)
n/N	relative sunshine duration
N	maximum possible duration of sunshine or daylight hours (hour)
N	nitrogen
N_2	diatomic nitrogen
NH_3	ammonia
NH_4^+	ammonium
NLATQ	nitrate in later flow (kg N ha^{-1})
NO_2	nitrogen dioxide
NO	nitrogen oxide
N_2O	nitrous oxide

NO_2^-	nitrite
NO_3^-	nitrate
NO_3-N	nitrate nitrogen
NO3GW	nitrate transported into main channel in the groundwater loading from the HRU (kg N ha^{-1})
NO3L	nitrate leached from the soil profile (kg N ha^{-1})
NO3_OUT	nitrate transported with water out of reach during time step (kg N)
NPERCO	nitrogen percolation coefficient
NSURQ	nitrate in surface runoff (kg N ha^{-1})
orgC	soil organic carbon content (%)
ORGN	organic nitrogen
ORGN_OUT	organic nitrogen transported with water out of the reach during time step (kg N)
ORGP	organic phosphorus
ORGP_OUT	organic phosphorus transported with water out of the reach during time step (kg P)
P	air pressure (kPa)
P	phosphorus
PO_4^{3-}	ortho-phosphorus
PET	potential evapotranspiration
pH	a measure of the acidity or alkalinity of a solution
PHOSKD	phosphorus soil partitioning coefficient
PPERCO	phosphorus percolation coefficient
ppm	parts per million
q	flow (m^3 s^{-1})
Q10	probability of 10% exceedance of river flow
R^2	fraction of the total squared error
r_a	diffusion resistance of the air layer (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)
ROT	rotation
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
Stdev	standard deviation
SURLAG	surface runoff lag coefficient
t ha^{-1}	tonnes per hectare

USLE_P	USLE crop management factor
$\mu g\ l^{-1}$	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 $°C^{-1}$)
λ	latent heat of vaporization (MJ kg^{-1})
λE	latent heat flux density (MJ m^{-2} d^{-1})

ABBREVIATIONS AND ACRONYMS

ABC	Agro Business Consultants Ltd
ABFS	Automated Base Flow Separation
ACCELERATES	Assessing Climate Change Effects on Land use and Ecosystems
ADAS	Agricultural Development and Advisory Service
AGNSP	Agriculture Non-Point Source model
ANSWERS	Areal Non-point Source Watershed Environmental Response Simulation
AnnAGNPS	Annualized Agricultural Non-point Source Pollution Model
AONB	Area of outstanding Natural Beauty
ATEAM	Advanced Ecosystems Analysis and Modelling
AVSWAT	SWAT Arc View-GIS based interface
AWC	Available Water Capacity
BADC	British Atmospheric Data Centre
BASINS	Better Assessment Science Integrating point and Nonpoint Sources
BFI	Base Flow Index
BMP	Best Management Practices
BOD	Biochemical Oxygen Demand
CAN	Calcium Ammonium Nitrate
CAP	Common Agricultural Policy
CASC2D	CASCade of planes in 2-Dimensions
CBOD	Carbonaceous Biochemical Oxygen Demand
CEH	Centre of Ecology and Hydrology
CLC	Corine Land Cover Map
CN	Curve Number
CORINE	Coordination of information on the environment
CREAMS	Chemicals, Runoff and Erosion from Agricultural Management System
CSF	Catchment sensitive farming
CWA	Clean Water Act
DEFRA	Department for Environment, Food and Rural Affairs
DEM	Digital Elevation Model
DM	Dry Matter
DO	Dissolved Oxygen
Dr.	Doctor
DTI	Department of Trade and Industry
DWF	Dry Weather Flow
DWSM	Dynamic Watershed Simulation Model
EA	Environment Agency
EC	European Community
ECSFDI	English Catchment Sensitive Farming Delivery Initiative
EDL	Edinburgh Data Library
EEC	European Economic Community
EPA	United States Environmental Protection Agency
EPIC	Erosion-Productivity Impact Calculator

EQS	Environmental Quality Standard
ESA	Environmentally Sensitive Area
ET	Evapotranspiration
EU	European Union
EURURALIS	A scenario study on Europe's rural areas to support policy discussion
FAO	Food and Agriculture Organisation of the United Nations
FAPRI	Food and Agricultural Policy Research Institute
GIS	Geographic Information System
GLEAMS	Groundwater Loading Effects of Agricultural Management Systems
GQA	General Quality Assessment
GS	Global Sustainability
HD	Habitat Directive
HOST	Hydrology of Soil Type
HRU	Hydrological Response Unit
HSPF	Hydrologic Simulation Program-Fortran
ICM	Integrated Catchment Management
IHDM	Institute of Hydrology Distributed
INTERREG IIIB	European Union Interreg Programmes for Strand B – North West Europe, Atlantic Area, and North Sea Area
JNCC	Joint Nature Conservation Committee
KINEROS	Kinematic Runoff and Erosion Model
LAI	Leaf Area Index
LandIS	Land Information System
LU	Livestock Unit
m	maize/corn
M	Measured
MAFF	Ministry of Agriculture, Fisheries and Food
MIKE-SHE	Generalised River Modelling Package - Systeme Hydroloque Europeen
MUSLE	Modified Universal Soil Loss Equation
NAEN	Natural Area by English Nature
NATMAP	The Digital National Soil Map of England and Wales
NDGMT	North Down Grassland Management Team
NGR	National Grid Reference
NEC	National Emission Celings
NRCS	Natural Resources Conservation Service (United States)
NSRI	National Soil Resources Institute
NVZ	Nitrate Vulnerable Zone
P	Phosphorus
PBIAS	Percentage Bias
PET	Potential Evapotranspiration
PRELUDE	PRospective Environmental analysis of Land Use Development in Europe
PRMS	Precipitation - Runoff Modelling System
QUAL2E	Enhanced Stream Water Quality Model
RE	Regional Enterprise

RegIS	Regional Climate Change Impact and Response Studies
rg	rye grass
RMSE	Root Mean Square Error
sa	set aside
SAC	The Scottish Agricultural College
SAC	Special Area of Conservation
SCS	Soil Conservation Service
SEISMIC	Spatial Environmental Information System for Modelling the Impact of Chemicals
SIMCAT	Statistical water quality model - Environment Agency (UK)
SMD	Soil Moisture Deficit
SSSI	Site of Special Scientific Interest
STW	Sewage Treatment Works
SWC	Soil water content
SWAT	Soil Water Assessment Tool
SWCS	The Soil and Water Conservation Society
TSP	Triple Super-phosphorus
UK	United Kingdom
US	United States
USA	United States of America
USDA	United States Department of Agriculture
USGS	United States Geological Survey
USLE	Universal Soil Loss Equation
WaSIM-ET	Water Balance Model - Evapotranspiration
wb	winter barley
WFD	Water Framework Directive
WTO	World Trade Organisation
WTW	Wastewater Treatment Works
wus	consumptive water use file
ww	winter wheat

1 INTRODUCTION

1.1 BACKGROUND

In many United Kingdom and European rivers high nutrient concentrations, specifically nitrate and phosphorus are causing deterioration in habitats and water quality. In 2005, 51% of the rivers in England and Wales had high concentrations of phosphorus (greater than 0.1 mg l^{-1}) and 28% of rivers had high concentrations of nitrate (greater than 30 mg l^{-1}) (Environment Agency, 2006-a). These nutrients, whilst essential for plant growth can, when used in excess, be lost to the river system through surface or subsurface flow routes. Their higher concentrations in surface water can cause an increased growth of individual species like algae, a decrease in dissolved oxygen and consequential eutrophication (Ash et al., 2005) and subsequent reduction in numbers of species.

The nutrients in the river Axe come from both non-point (diffuse) and point sources. The diffuse sources are predominantly agricultural while point sources are a mix of discharges from sewage treatment works (STW) and industry. River water quality (defined here as nitrate and phosphorus concentrations) is a result of a complex mixture of continuous and intermittent sources. Historically it has been easier to identify and regulate point sources. However it is sometimes considered uneconomic to restrict smaller discharges. Phosphorus striping is not generally introduced for smaller Sewage Treatment Works (less than 10,000 population equivalent), which are dominant in many rural areas (Wheater and Daldorph, 2003; Neal, 2005).

The Water Framework Directive (Directive 2000/60/EC) requires new ecological standards for rivers, lakes and coastal waters by 2015 (European Union, 2000) and together with the Nitrate Directive (Directive 91/676/EEC), Habitats Directive (Directive 92/43/EEC) and Urban Wastewater Treatment Directive (Directive 91/271/EEC) means that nitrate and phosphorus concentrations must be addressed. It is likely that this will be achieved through a focus on diffuse sources to augment the existing constraints on point sources. Within the perspective of the WFD, the UK Department for Environment, Food and Rural Affairs (DEFRA) identified a number of

catchments as priority sites for the catchment sensitive farming (CSF) initiative (DEFRA, 2002-a). CSF aims to introduce a number of mitigation methods into farming practice to reduce contamination of rivers and water bodies.

This thesis is focused on the river Axe catchment in south-west England with a mixture of diffuse and point sources of pollution which is one of these priority sites, as intensive dairy farming and cultivation of high risk crops (maize) for fodder, cause problems with enhanced diffuse nutrients and especially phosphorus levels in the river. The Environment Agency identified the river Axe catchment as suffering from high phosphorus levels in particular (Ash et al., 2005), although sediment and nitrate are also issues.

1.2 AIM AND OBJECTIVES

High concentrations of nutrients in river water can lead to a reduction in biodiversity which is a prime concern in the River Axe catchment. This research is undertaken in the light of the Water Framework Directive (WFD), Nitrate Directive (ND), Habitats Directive (HD), Special Areas of Conservation (SAC), the English Catchment Sensitive Farming Delivery Initiative (ECSFDI), and the Blackdown Hills Environmentally Sensitive Area (ESA) scheme. Therefore the aim of this research is to identify the impacts of different land use management scenarios on diffuse source nutrients for delivery of water quality improvements in the River Axe catchment.

The objectives are:

- to analyse the present catchment land use, land management and diffuse and point source nutrient situation,
- to define a modelling approach and its advantages and disadvantages,
- to collect, arrange and analyse data required and estimate missing data,
- to apply a model to the catchment for current conditions and carry out calibration, validation and evaluation of the model performance,
- to define possible land management scenarios,
- to use the selected model to investigate the impacts of these defined land management scenarios on nutrient loads to the river,
- to recommend possible mitigation measures to deliver water quality improvements.

2 LITERATURE REVIEW

2.1 RIVER ECOSYSTEM

Surface waters are an important source of water for drinking and irrigation, food and energy and a valuable transport route for human kind. In most parts of the world river ecosystems are crucial features in the everyday life of humans. But rivers and their banks are not populated only by the human race. The river fauna and flora is known as one of the richest ecosystems with a wide range of different varieties. The influence of humankind on water bodies increased dramatically during the industrial revolution in the 18th century with a growing world population and increasing water abstraction and industrial and municipal discharges. In 1962, after the book "Silent Spring" (Carson, 1962) was published, independent environmental movements raised the alert and pointed out the highly impacted state of surface waters, which resulted in declining river quality and even the extinction of some species.

With developing industry and rising population in the 20th century, agriculture was forced to increase production of food. This was possible only with higher rates of fertiliser and pesticide usage. Inevitably in some places, their application at a rate or time above the crop requirements did not result only in higher crops yields but also in a threat to the surface waters. A high livestock density in one location or land, over treated with fertilisers in another, was, and in some parts of the world still is, an ecological time bomb. Natural processes act to disperse concentrated nutrients, and the logical consequence is movement of nutrients in the water flows under gravity to the nearest water body (river, lake and groundwater). The results of such processes are high concentrations of diffuse nutrients, such as nitrate (NO_3^-) and ortho-phosphorus (PO_4^{3-}), in surface water bodies, which in many cases leads to eutrophication.

However, agriculture isn't the only polluter of surface waters. Wastewater treatment works (WTW) and industrial discharges were in the past major polluters of the rivers and in most cases, globally, they still are. The costs for further lowering of nutrient concentrations in WTW discharges are, in many cases, higher than the recognised

benefits of this action. In recognition of this fact the emphasis has in recent years turned back from point sources to the diffuse source polluters.

River ecology is also affected by flow regime, as velocity in the upper reaches of rivers may be too fast to allow micro-organisms, algae or flowering plants to develop, but in low and meandering reaches with many wetlands, where flow velocity is slow, a noticeable range of phytoplankton, algae or flowering plants may develop (Gibson, 1997). Knowing these linkages, even more concern for water quality in river ecosystems is justified, especially if water is used in public water supply. Peterson et al. (1985) have characterized diffuse source pollution problems as among the most pervasive, persistent and diverse water quality problems in the USA and this could be transferred to the world scale. But human needs are not the only thing that must be taken into account; wildlife is also an important part of the river ecosystems as it can serve as a bioindicator for water quality and is part of the healthy ecological functioning of aquatic systems. Different designations and sustainable development of river ecosystems can help in their survival and existence for future generations.

Throughout the world governments have recognised the problem of polluted rivers. In the European Union (EU) public discussion about improvement of water quality in Community surface waters began in 1975 with enforcement of standards for rivers and lakes used for drinking water abstraction. In 1988 a Ministerial Seminar on Community Water Policy was held in Frankfurt, and reviewed existing and suggested improvements to the water legislation (European Union, 2000). From then to date, several directives have been adopted including the Urban Waste-water Treatment Directive (91/271/EEC), the Nitrate Directive (91/676/EEC), the Integrated Pollution Prevention and Control Directive (96/61/EC) and finally the Water Framework Directive (2000/60/EC).

The objective of the Nitrate Directive (91/676/EEC) is to reduce water pollution by nitrate from agriculture to acceptable levels from 2003 onwards (European Union, 1991-b). The nitrate and ammonium concentrations in ground water and potable water, as defined in this Directive, may not exceed 50 mg NO_3^- l^{-1} and 0.5 mg NH_4^+ l^{-1} as maximum allowable concentrations. Concentrations of less then 25 mg NO_3^- l^{-1} and

0.05 mg NH_4^+ l^{-1}, as guide level, are recommended. Emission reduction goals for ammonia (NH_3) have been agreed by European member states in the National Emission Ceilings (NEC) Directive (2001/81/EC). Some of the member states, such as the Netherlands has committed itself to a reduction of losses of gaseous ammonia of 22% by 2010, compared to emissions in 1999 (Aarts, 2003). Such emissions are mainly due to dairy farming (Jarvis, 1999). The most important act for preserving water resources in the EU was implementation of the Water Framework Directive (WFD) (2000/60/EC) in the year 2000 (European Union, 2000). The WFD supercedes all previous directives and any other relevant legislation concerning water quality in the EU. The aim of the WFD is to contribute to the progressive reduction of emissions of hazardous substances to the waters and achieve 'good ecological status' of groundwater and surface water at the latest 15 years from the date when the WFD entered into force, that is by 2015. This directive aims to reduce water pollution by any contaminant which prevents good ecological status, but particular emphasis is given to nitrogen (N) as nitrate, ammonium, organic N, phosphorus (P), heavy metals and pesticides.

The United Kingdom (UK) is, as a member state of EU, obligated to enforce all its legislation. For that reason the UK government has adopted the WFD and Nitrate Directive (ND) into their legal system. A number of initiatives have been designed to address the WFD and ND requirements, such as the England Catchment Sensitivity Farming Delivery Initiative (ECSFDI), Nitrate vulnerable zones (NVZ) and others (DEFRA, 2005; DEFRA, 2007-a). In total 68 NVZ-s were designated in 1996 in England and Wales covering 600,000 ha (Davies, 2000) and in Scotland, 6 NVZ-s have been established during the period from 1996 to 2003 (Macgregor, 2006). Implementation of actions within areas defined as NVZ-s also requires farmers to adopt different management practices and techniques. For easier and more sufficient implementation of environmental legislation in to real life governmental agencies also help polluters through different national grant schemes.

England and Wales are no exception to this story as in 2005, 51% of the rivers had high concentrations of ortho-phosphorus (greater than 0.1 mg l^{-1}) and 28% of rivers had high concentrations of nitrate (greater than 30 mg l^{-1}) (Environment Agency, 2006-a).

2.2 DIFFUSE SOURCE NUTRIENTS

Sources of water pollution are classified as point sources and non-point (diffuse) sources. Point sources are mostly identified with industry and waste (sewage) water treatment plants where pollutants are delivered through a pipe - discharge point (Wolfe, 2000). Diffuse pollution is associated with numerous dispersed sources, such as sediment, manures, fertilizers and pesticides and have their source mainly in agriculture, although urban diffuse pollutants (roads, gardens, parks) are present too. They can be transported by surface flow, lateral subsurface flow and can percolate to groundwater and from there to rivers. Mobilization and delivery of diffuse pollutants is often dependent on weather conditions (rainfall-runoff), and can be influenced by soil type, land management and topography (Campbell and Edwards, 2000).

The most important diffuse source nutrients are nitrogen (N) in the forms of ammonia (NH_4^+), nitrate (NO_3^-), and phosphorus (P) in the ortho-phosphorus (PO_4^{3-}) form. Surface waters can be impacted by runoff from feedlots, manure storage, from manure and fertiliser land application sites, and from grazed pastures. Sediment is an important source and vehicle for the transfer of soil-bound chemical pollutants from diffuse source areas to waterways (Nearing et al., 2000). Soil erosion is, according to the USDA (1989), the source of 80% of total phosphorous and 73% of total Kjeldahl nitrogen in the waterways of the USA. Where animals have direct access to streams, their excreta may enter directly to the water. Groundwater is impacted in general by nitrate leaching through the soil profile from manure and fertiliser application sites and also by leaching from manure storage basins and lagoons (Ritter, 2000-b).

Sharpley and Withers (1994) suggest that on average, agriculture contributes 40% of the total external phosphorus loading of surface waters in the UK. A more recent study from White and Hammond (2006) estimated total phosphorus load contributions to waters in the UK as 28.3% from agriculture, 60.7% from household, 4.6% from industry and 6.5% from background sources. The latest research (Ash et al., 2005) on the river Axe, before WTW phosphorus stripping was introduced, showed that diffuse sources contributed over 52% of total phosphorus load. Hunt et al. (2003) estimated

total nitrogen load contribution to water in England and Wales to be 60.6% from agriculture, 32.1% from STW, 4.1% other land, 1.8% industrial, and 1.4% other.

Any amount of phosphorus and nitrogen in the water exceeding the minimum required for algal growth can lead to accelerated eutrophication and consequentially to reduction of aquatic life and species diversity. Eutrophication is defined as an increase in the nutrient status of natural waters that causes growth of algae or other vegetation, reduction of dissolved oxygen, increased turbidity, and degradation of water quality (Ritter, 2000-b). It is a naturally occurring process characterized by biological activity, but it is often accelerated by pollution from human activities. The eutrophication threshold for most P-limited systems is from 0.010 to 0.100 mg P l^{-1} and for N-limited systems from 0.5 to 1.0 mg N l^{-1} (Mason, 1991).

2.2.1 Nitrogen (N)

Nitrogen is one of the essential nutrients for living organisms and an important factor limiting crop yield. A considerable amount of research has been done to achieve a better understanding of processes controlling the nitrogen cycle (Figure 2.1) in various ecosystems. The nitrogen cycle is very complex because it can occur in many valance states. Certain processes occur only aerobically or only anaerobically, and they are regulated to a large extent by microbial processes in complex soil structure under non steady-state conditions (Ritter, 2000-b).

The main sources (Table 2.1) of diffuse nitrogen originate in livestock wastes (manure, sludge) and mineral fertilisers. Other sources of nitrogen are mineralization of nitrogen containing compounds in soil organic matter and plant residues, and atmospheric deposition of ammonia and ammonium which can annually amount to less than 10 kg N ha^{-1} in Ireland (Sherwood and Tunney, 1991) and up to 50 kg N ha^{-1} in areas with intensive livestock in the Netherlands (Verbruggen et al, 2003). Biological fixation of atmospheric nitrogen can supply substantial quantities of mineral nitrogen (100 to 300 kg N ha^{-1} $year^{-1}$) in symbiosis with leguminous species such as white clover (Masterson

and Murphy, 1983). Small quantities of nitrogen are also fixed in soil by free-living bacteria (Whitehead, 1995).

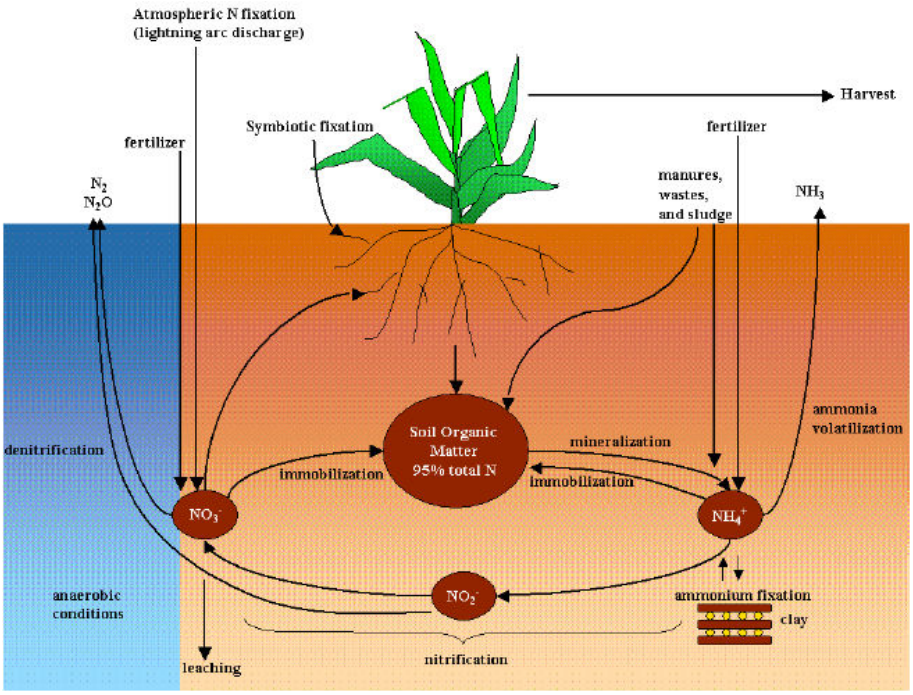


Figure 2.1: The nitrogen cycle (Source: Neitsch et al., 2005).

Table 2.1: Sources and sinks for the nitrogen budget (Adapted from: Campbell et al., 2004; Connell, 2005, Koo, 2006).

Sources	Sinks
Leachate from leaves and miscellaneous debris	Water flow
Agricultural (cropland) and drainage	Groundwater recharge
Animal waste runoff	Fish harvesting
Marsh drainage	Weed harvesting
Runoff from uncultivated and forest land	Volatilization (of NH_3)
Urban run-off	Denitrification
Domestic waste effluent	Irreversible sediment deposition of detritus
Natural groundwater	Sorption of ammonia onto sediments
Subsurface agricultural and urban drainage	Crop uptake
Nitrogen fixation	
Atmospheric deposition	
Sediment leaching	

Nitrogen mineralization is the processes, mainly carried out by microorganisms, through which organically bound nitrogen is converted to ammonium nitrogen (NH_4^+). Several biotic and abiotic factors and processes compete for very soluble and available

NH_4^+ (e.g., nitrification to NO_3^- and plant uptake), therefore levels of NH_4^+ are very low in cropped agricultural soils; in many cases below 5 mg kg^{-1} soil (Bowen, 1981). But ammonium ions are not very mobile, because they are bound to cation exchange sites within soils (Humphreys et al., 2003). For better understanding of mineralization and immobilization the general relationship of carbon/nitrogen (C:N) ratio is crucial. When C:N > 30:1 immobilization (decrease in NH_4^+ and NO_3^-), C:N < 20:1 mineralization (increase in NH_4^+ and NO_3^-), 20:1 ≤ C:N ≥ 30:1 (no change - equilibrium) (Neitsch et al., 2005).

Forms of reactive and highly water soluble nitrogen that can cause water pollution are ammonia (NH_3), ammonium (NH_4^+), nitrite (NO_2^-), and nitrate (NO_3^-). The nitrate ion (NO_3^-) is usually the dominant form in which nitrogen occurs in surface and ground waters (Burt and Haycock, 1992). It is also generally present in warm and moist UK soils and is liable to leach at any time of the year, especially in late autumn and early winter when precipitation exceeds evaporation and when land is bare and crop uptake is low (Davies, 2000). Plants can uptake nitrogen from soils in two different forms as ammonium ion (NH_4^+) or as nitrate ion (NO_3^-), which is more prevalent and therefore taken up in large amounts. The total amounts of nitrogen taken up by plants vary considerably depending on the type of crops, soil type, climate and other environmental conditions.

Nitrogen compounds can be lost from the agricultural land system also as gases during ammonia volatilization from ammonium (NH_4^+) to ammonia (NH_3) and during denitrification from nitrate (NO_3^-) to nitrous oxide (N_2O), nitrogen oxide (NO), nitrogen dioxide (NO_2), and diatomic nitrogen (N_2).

2.2.2 Phosphorus (P)

Phosphorus is an essential element for all forms of life and has many important roles in natural ecosystems and in agriculture. Phosphorus inputs for plant growth are necessary to maintain profitable crop production. With human activities, such as intensive agriculture, there exists the possibility for over-enrichment of aquatic ecosystems (Table

2.2). Haygarth et al. (1998) found that generally, less than 2% of the annual total phosphorus input (mineral and organic fertilisers, cow excrement) is lost from a dairy farm. This represents only a small proportion of all applied phosphorus but it can have significant impact as phosphorous inputs can increase the biological productivity of aquatic ecosystems and can lead the water body to be in a eutrophic state (Campbell and Edwards, 2000).

Table 2.2: Phosphorus sources and environmental problems. (Source: Campbell et al., 2004)

Sources	Problems
Soil erosion	Eutrophication of freshwaters:
Subsurface flow	Ecological degradation
Agricultural fertilizers, manure	Increased potable water treatment cost
Contamination of urban run-off	Nuisance algal growth
Sewage treatment works	

Ortho-phosphorus (PO_4^{3-}) is a major water pollutant and plays a significant role as the limiting nutrient in eutrophication. Its concentration is important because so little is required for biomass production. The ratio of nutrients in biomass production in aquatic ecosystems is 105:15:1 - carbon (C): nitrogen (N): phosphorus (P) (Redfield, 1934; Uhlmann and Albrecht, 1968). That means that adding one atom of P has greater effect than adding either N or C, even assuming that they were limiting. Considering this fact it is not surprising that a small concentration of 0.01 mg P l^{-1} is sufficient for accelerated algae growth. Further increases in P concentration over 0.05 mg P l^{-1} tend to produce many problems like deoxygenation, and concentrations over 0.1 mg P l^{-1} can lead to obvious eutrophication (Gibson, 1997).

Soil phosphorus is present in numerous different forms (Figure 2.2), many of which are relatively stable and are not readily available for plants or soluble in water (Campbell and Edwards, 2000). Soil phosphorus exists in inorganic and organic forms. Processes that describe the phosphorus cycle are functions of soil, weather, and crop variables e.g., adsorbtion/desorbtion, precipitation/dissolution, immobilization/mineralization, and plant uptake/plant decomposition. Organic phosphorus forms mineralize and replenish to inorganic through microbial activity. All forms of inorganic phosphorus (Fe, Al phosphates in acid soils and Ca phosphates in alkaline soils) in soils are extremely

insoluble. In normal conditions inorganic phosphorus is converted from mineral forms to bioavailable and soluble forms by dissolution through the weathering processes. Phosphorus fixation or precipitation is a chemical reaction, which may hold soluble and bioavailable phosphorus forms in place in the soil, meaning they are not available for plants.

Phosphorus is adsorbed to the soil fraction that is most susceptible to erosion (clays, oxides of Fe and Al). Therefore, adsorbed phosphorus can enter surface waters by sediment transfer; desorbed or solution forms are transferred in runoff independently of sediment. The pH is also a critical factor determining adsorption. Although different minerals adsorb P maximally at different pH-s, in most soils P availability is maximal at 5.5 to 6.5 (Eviner et al., 2000). The most obvious aspect of phosphorus availability for plants and losses from land is that it is controlled by highly site-specific processes (Morgan, 1997). This was confirmed by Pionke et al. (1997); they found that 98% of the sediment bioavailable phosphorus loss is from 6% of the catchment and most of the dissolved phosphorus loss is from 11% of the catchment area.

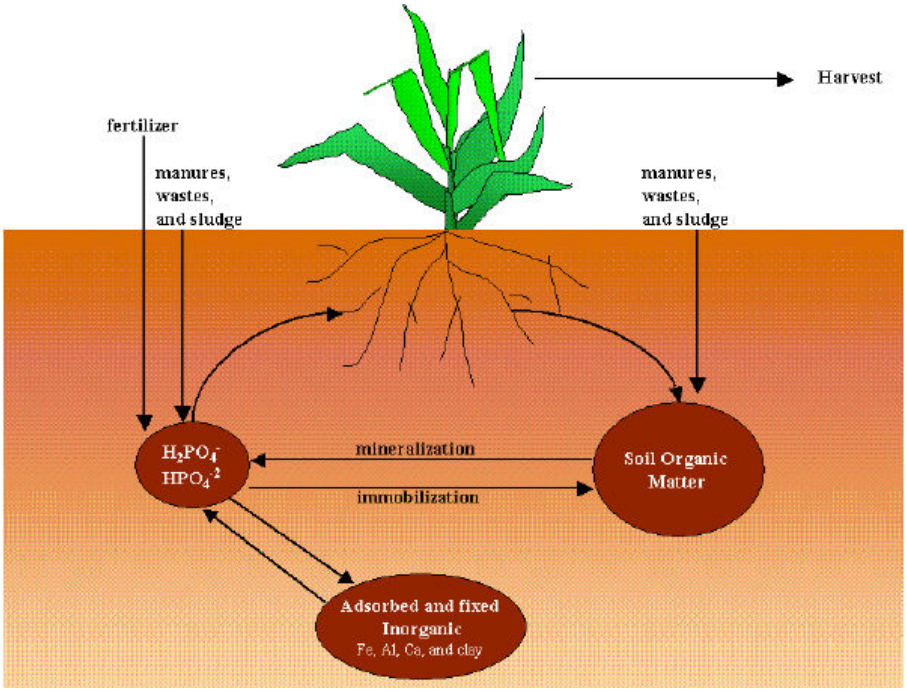


Figure 2.2: The phosphorus cycle (Source: Neitsch et al., 2005).

2.3 DIFFUSE NUTRIENTS TRANSFER

Diffuse nutrient pollution and transfer routes are closely connected to the hydrological cycle managed by water and energy from the sun. Diffuse nutrients (nitrogen and phosphorus) are transferred to waters by surface runoff (N, P) or subsurface flow (N, P) to surface water and by leaching to groundwater (N). In addition, groundwater feeds streams, so pollutants can reach surface water from the groundwater (Wolfe, 2000). On some occasions nutrients can be added by direct input like fertiliser if the product is thrown by the spreader directly into the water or if animals have direct access to the water and may defecate or urinate into water (Campbell et al., 2004).

2.3.1 Surface runoff

Surface runoff is a complex, variable process which occurs when the infiltration capacity of the soil is exceeded by the rainfall rate. Surplus rain (in excess of infiltration) accumulates on the soil surface and runs off when the depth of ponding and other surface conditions cause the water to flow. Runoff travels across the land surface, increasing and decreasing in flow velocity and changing course depending on slope, vegetation, surface roughness, and other surface characteristics. It can infiltrate as it flows (transmission losses) or previously infiltrated water can re-emerge to join the surface flow. The amount of runoff depends on infiltration, interception, evapotranspiration (ET), and surface storage (Wolfe, 2000). It may contain high concentrations of sediment particles of different sizes and diffuse nutrients.

Catchment characteristics influencing surface runoff are soil properties, land use, vegetation cover, moisture condition, size, shape, topography, orientation, geology, management practices, and channel characteristics (Wolfe, 2000). For example canopy interception was found to be a very important element in decreasing surface runoff. Well developed forest canopy can intercept about 10% to 20% of the annual rainfall (Linsley et al., 1988).

Surface runoff may occur as sheet-type flow with small depths of flow and slow velocities (less than 0.3 m s^{-1}). However, more commonly, flow is concentrated and rill and gully erosion can transfer considerable volumes of water, sediment and diffuse nutrients in dissolved forms and adsorbed to the sediment (Wolfe, 2000). Fine clay particles or organic matter, which have large surface areas and relatively high electrical surface charge, are the major adsorbents and vehicles for transfer of a strongly adsorbed inorganic nutrient like phosphorus (Nearing, 2000). Phosphorous and nitrogen surface transfer factors are precipitation (in the form of rainfall total depth and duration), soil texture (fine texture soil favours surface runoff), soil cover (low cover promotes high surface runoff), soil moisture (high soil moisture at the time of the rainfall event favours the occurrence of runoff) (Campbell and Edwards, 2000).

Methods of fertilizer application, land use management and season of application can significantly affect nutrient losses in surface runoff. Spreading manure on frozen or snow-cowered ground increases losses by surface runoff and as Hensler et al. (1970) reported, up to 20 % of nitrogen was lost from manure applied to frozen tilled soil.

2.3.2 Infiltration and lateral subsurface flow

Infiltration is a process, where water and dissolved nutrients enter from the surface into the soil profile. The lower the soil moisture, the higher is the infiltration rate (Wolfe, 2000) which is affected by amount of water available for infiltration, rainfall intensity, surface conditions, including roughness, vegetation characteristics, and surface sealing. Surface sealing can reduce infiltration rates by 25 to 35% for sand loam to silty clay loam and up to 75% for a clay loam soils (Duley, 1939). Other studies report for clay loam soils reduction in infiltration rates by 20 to 30% (Mannering, 1967) and up to 50% (Edwards and Larson, 1969). Lateral flow will be significant in areas with soils having high hydraulic conductivities in surface layers and impermeable or semi-permeable layers at shallow depth in the soil profile (Neitsch et al. 2005).

Infiltration and lateral flow are processes mainly connected with nitrogen, but in certain conditions phosphorus can also be infiltrated and moved with subsurface flow. Two

conditions have to be fulfilled before nutrient infiltration starts. The NO_3^- or PO_4^{3-} levels in the soil solution have to be sufficiently high and downward movement of water has to be high enough to displace the available nutrients below the rooting zone (Ritter, 2000-a). The work of Shirmohammadi et al. (1997) in the Piedomont physiographic region showed that lateral flow plays a major role in NO_3^- loading to streams. Infiltration and subsurface phosphorus transfer can occur in many sandy soils with extremely low phosphorous adsorption capacity so phosphorus added to the soils often moves readily in water (Graetz and Nair, 1995). Phosphorus over fertilisation can also increase the potential for losses to drainage water because no soil has an infinitive capacity for phosphorus adsorption and precipitation. Dils and Heathwaite (1999) suggest that subsurface flow may be a significant contributor of phosphorus in grassland systems relative to arable catchments. This is due to enhanced infiltration through high surface roughness in grassland catchments, coupled with cultivation practices which encourage macropore formation.

Drainage systems are mainly constructed for draining the water from agricultural land on clay-based soils. Well-planned and well-managed drainage systems change the hydrologic relationships on the land (Fausey et al., 1995). Subsurface drainage can reduce erosion and the amount of surface runoff and peak rate of discharge (Ritter and Shirmohammadi, 2000). Drainage water quality studies report that concentrations of nitrate (NO_3^-) are greater in subsurface drainage than in surface runoff, and that ammonium (NH_4^+) and phosphorus (P) concentrations are greater in surface runoff than in subsurface drainage (Baker and Johnson, 1977 cited in Ritter and Shirmohammadi, 2000). Studies showed that nitrate concentrations of 10 – 20 mg NO_3^- l^{-1} are common for USA subsurface drainage (Neely and Baker, 1989) and that peak concentrations in UK drainage systems can reach 30 – 50 mg NO_3^- l^{-1} in winter time (Rose et al., 1991).

2.3.3 Groundwater leaching

Groundwater and surface water are interrelated through recharge and discharge. Groundwater is recharged from movement of soil moisture through the vadose (unsaturated) zone to the saturated zone; recharge of groundwater can also occur from

surface water bodies. Natural discharge from groundwater occurs through springs, spring-fed lakes, wetlands, stream channels and oceans. Discharge also occurs from pumping for a variety of users, among them being agriculture (Wolfe, 2000). These interactions between groundwater, soil and surface waters are reflected in diffuse nutrient pollution problems. There are three ways of groundwater interaction with streams, (1) gaining or (2) losing water from groundwater through the streambed, or they can do both (3), gaining in some reach areas and losing in other reach areas (Winter et al., 1998).

Flow in groundwater systems is usually slow and depends on hydraulic conductivity. Typical velocities may range from less than 1 cm $year^{-1}$ in tight clays or more than 100 m $year^{-1}$ in permeable sand and gravel (Novotny and Olem, 1994). The normal range of groundwater velocities are 1.5 m $year^{-1}$ to 1.5 m day^{-1} (Todd, 1980). Most vulnerable for diffuse source pollution are unconfined shallow aquifers, because changing meteorological conditions strongly affect surface water and groundwater patterns in such water bodies. Several researchers have confirmed significant denitrification at locations where oxygen is absent or present in very low concentrations in aquifers with silt and clay confining beds and in riparian areas near to the stream. For example, results from McMahon and Bohlke (1996) showed that nitrate (NO_3^-) load in discharging groundwater was 15 - 30% lower than the load that would have resulted from the discharge of unaltered groundwater from the recharge area.

2.4 CATCHMENT LAND MANAGEMENT

For good and effective catchment management, land use for a given area has to be defined first. This is fundamental to assess diffuse source pollution primarily because it denotes the activity on land (or use of the land) that generates a pollutant of concern (Clapham et al., 1999). Agricultural land can be sub-divided into cropland, grazed pastures, non-grazed pastures, and irrigated or drained land. The importance of agricultural land use in Europe is clear as 53% of the European land surface is classified as agriculture and management of this land has wide range impacts on the environment

and especially on the nutrient dynamics of water resources, and biological diversity (Rounsevell et al., 2003).

With changing land use management, especially agricultural, it is possible to reduce the main sources of nitrogen and phosphorus losses to the water system. There is no single solution that would achieve required standards, because the pollution loads come from many sources and directions affecting surface water and groundwater in multiple ways (Campbell et al., 2004).

2.4.1 Best management practices

To minimize influences of pollution resulting from agricultural activities, best management practices (BMP-s) can be implemented. They should not only be based on whether the practices will provide pollutant reductions that will achieve water quality goals, but also on whether implementation of the practices is economically feasible for the parties involved (Mostaghimi et al., 2000). Importance of practicability is shown also in the definition of a BMP by The Soil and Water Conservation Society (SWCS) (1982) as "*a practice or combination of practices that are determined by the state or designated area wide planning agency to be the most effective and practicable (including technological, ecological, and institutional consideration) means of controlling point and non-point source pollutants at levels compatible with environmental quality goals*".

Best management practices have to be focused on reduction of surface runoff, on reduction of the generation of sediments in surface runoff, and availability of nutrients and pesticides. Therefore solutions consist of many different approaches and managements on a multi–criteria basis as presented in Table 2.3.

Table 2.3: BMP-s measures that offer general benefits for nitrogen, phosphorus and suspended solids (Source: Campbell et al., 2004)

BMP	P	NO_3^-	NH_4^+	Suspended solids
Waste management plans	+	+	+	(+)
Nutrient budgets	+	+	(+)	-
Buffer strips	(+)	-	?	(+)
Contour ploughing	+	-	?	+
Farm ponds or reedbeds	+	+	+	+
Composting	?	+	+	-
Slurry digestion	?	+	+	-
Reducing livestock density	+	+	+	+

+ (Positive), - (Small or no benefits), ? (No data)

When using best management practices to control nitrogen and phosphorus diffuse pollution it is very important to take into account all sources of nutrients (organic and mineral fertilizers, legumes, and soil organic matter) to understand the amount of nutrient already present in the environment. In accordance with the EU Nitrate Directive (91/676/EEC) regulations, the maximum amount of animal manure that can be applied to farmland each year should not exceed 170 kg N ha^{-1}, including excreta from grazing animals. To meet this requirement, different BMP measures were applied for UK Nitrate Vulnerable Zones (NVZ) to reduce nitrate leaching. These included conversion to permanent grassland, limiting manure to 170 kg N ha^{-1} $year^{-1}$ and applying in spring (no autumn nitrogen), avoiding fertilisation in high risk areas and times, early sowing of autumn sown crops, keeping a green cover over winter, reducing stock and grazing densities, avoiding poaching, developing nutrient management plans, soil tests, etc. (Burt and Haycock, 1992; Croll, 1994; DEFRA, 2002-b).

All BMP-s have their background in chemical and physical processes such as the C/N ratio which plays a very important role in the nitrogen cycle. For example, in mixed pastures of grasses and legumes, it is usually the legume leaf litter with a lower C/N ratio than the above-ground grass residues that contributes to net nitrogen mineralization during decomposition (Palm, 1991). Introduction of nitrogen-fixing legumes will not only provide an atmospheric N input to the system but also reduce immobilization of nitrogen and hence improve the general soil fertility (Urquiaga, 1993).

Crop selection can have a significant influence on phosphorus uptake (Table 2.4); corn takes up more than 2.5 times the amount of P as oats at typical yields (Campbell and Edwards, 2000). Of course uptake also depends on other factors like crop growth, temperature, soil moisture, soil pH and other nutrient availability.

Table 2.4: Typical phosphorous uptake of common crops (Source: Campbell and Edwards, 2000)

Crop	Yield (kg/ha)	P uptake (kg/ha)
Corn	11,300	50
Soybeans	3,460	26
Grain sorghum	8,400	39
Wheat	9,500	25
Oats	3,600	20
Barley	6,500	32
Alfalfa - Lucerne	18,000	59

Buffer strips

They are defined as planted or indigenous bands of vegetation that are situated between pollutant source areas and receiving waters to reduce transported pollutants from surface and subsurface runoff (Mostaghimi et al., 2000). They encourage slow percolation and long retention times, essential for nitrate removal by assimilation and denitrification (Haycock and Burt, 1993-a) and they reduce surface runoff velocities by vegetation roughness and the physical filtering effect of dense vegetation (Muscutt et al., 1993). The effectiveness of buffer strips depends on their slope and width, the lower the slope and the wider they are, the more effective they are. Kronvang et al. (2000) found that 29 m wide buffer strips, retained all sediment and P and in contrast for 0.5 m wide buffers, 38% of sediment and 68% of P from rill erosion passed through to the stream. Robinson et al. (1996) observed that 3.0 m wide buffer strips effectively removed up to 70% of the sediment load from cropland runoff. Dillah et al. (1989) observed sediment trapping efficiency for 4.6 m wide buffer strips of 53% to 86%. Buffers are expected to function up to 10 years, but without proper maintenance (regular harvesting of the biomass grown) may become a source of pollution because of the nutrient accumulation.

Other options are riparian buffer zones or ecotones; an area consisting of trees and shrubs that are located directly adjacent to permanent or intermittent water bodies. They reduce sediment loads and sediment-bound pollutants entering watercourses in run-off from agricultural land (Lowrance et. al, 1984; Peterjohn and Correll, 1984; Jacobs and Gilliam, 1985; Pinay and Decamps, 1988, Dillaha and Indamdar, 1997, Grill et al., 1997). Martin et al. (1999) found that two riparian headwater stream zones removed nearly 100% of the NO_3^- from subsurface lateral waters. Haycock and Burt (1993-b) suggests that near-stream land, such as undrained floodplains should be taken out of production in order to make rivers less sensitive to the pollution problems associated with modern intensive agriculture and land use changes within their catchment area.

Cover crops

Cover crops are grown during the time period between harvest and planting of the primary crop to provide soil cover and protection against soil erosion (Mostaghimi et al., 2000). Winter cover crops such as rye were found to offer the most potential for rapid nitrogen uptake. Nitrate concentrations on two Coastal Plain catchments were consistently lower when a rye cover crop was present than in previous years when no cover crops were present (Brinsfield and Staver, 1991). The importance of forage cover crops was also confirmed by Feng et al. (2005), as leaching of residual soil nitrogen was effectively reduced.

Nutrient management

Managing the rate, timing and placement of organic and mineral fertilizers to encourage maximum nutrient recycling and minimize nutrient runoff and leaching (Gitau, 2004) is good practice. For good nutrient management soil tests are needed to determine residual levels of available nutrients and the nutrient need of the crops for specific yearly yield goals has to be determined (Mostaghimi et al., 2000). Incorporating manure into the soil, either by tillage or subsurface injection, increases the amount of manure nitrogen available for crop uptake and can reduce the potential water pollution by nutrients in surface runoff, infiltration or leaching (Ritter, 2000-b; Osei et al., 2003). Incorporation

of dairy manure by chisel plowing reduced total phosphorus loss in surface runoff from corn 20 times compared with no-till areas receiving surface application (Mueller et al., 1984). Davies (2000) states that soil incorporation of surface applied slurries, decreases ammonia volatilization, but increases the potential for losses from leaching and denitrification. Winter application of manure is the least desirable, because frozen soil surface and an inactive plant canopy can result in nutrient loss and pollution through leaching and surface runoff (Ritter, 2000-b). Recycling of nitrogen under grazing management is a very inefficient process, particularly by grazing cattle or sheep if the stocking density is too high. Losses by surface runoff or leaching can be high, however grass has a relatively large capacity to take up nitrogen from soil compared to other crops; annual uptake of over 700 kg N ha^{-1} (Humphreys et al, 2001). Therefore under cutting management where all the grass is mechanically harvested, transported and fed to livestock indoors, little residual nitrogen remains in the soil under normal levels of fertilisation. Excreta collected and stored can be applied to grassland using low-emission techniques at optimum times of the year to ensure maximum utilisation of nitrogen (Humpreys et al, 2003).

Crop rotations

Typically defined as a planned sequence of annual and/or perennial crops, or planned grazing sequences. Rotational grazing is defined as the controlled harvest of vegetation with grazing or browsing animals. It is applied to prevent pollution that may result from high animal stocking densities on the land. Water quality impacts from pastured livestock areas depends on the stocking density, length of grazing period, average manure loading rate, manure spreading rate, manure spreading uniformity by grazing animals, and disappearance of manure with time (Sweeten and Reddel, 1995, Mostaghimi et al., 2000). Olness et al. (1975) found that continuously grazed pastures contribute at least four times more nitrogen and phosphorus in surface runoff compared with rotationally grazed pastures. A rotational crop system is not only important for reduction of pests but also for removal of nutrients in excess from the soil and erosion reduction. Planting forage crops in rotation with grain crops helps to remove the excess phosphorus and potassium and keep the three nutrients (N, P, K) in balance (Ritter,

2000-a, Campbell and Edwards, 2000). Burwell et al. (1975) found that surface runoff on loamy soils in Minnesota (U.S.) could be reduced from 23.8 to 3.3 kg N ha^{-1} by switching from continuous corn to hay rotation. In certain circumstances nitrogen losses from ploughed grassland are very high, especially in autumn. Temporary grassland in a rotational system does not cause high leaching when the length of the grass period is restricted to three or four years and it is ploughed in spring. Aarts (2003) suggests that diffuse nutrients after maize harvesting should not be a problem, because maize uptakes the majority of mineralized nitrogen from grass.

2.4.2 European Union Common Agricultural Policy

The European Union Common Agricultural Policy (CAP) has had a great influence (subsidies payments, intervention buying) on land use changes and consequently on nitrate problems (Addiscott, 2005). Introduction of regulations like milk quotas or restrictions like timing of manure application lead to more efficient use and reduction of manure and mineral nitrogen inputs in all European Union member states. Due to this nitrogen surpluses on Netherlands dairy farms in period 1986-2001 show a decreasing trend from about 400 kg ha^{-1} in the early 1980s to about 200 kg ha^{-1} in 2001. In the same period phosphorus surpluses have decreased from 31 kg ha^{-1} to 23 kg ha^{-1} (Aarts, 2003). The CAP is changing all the time and new reform suggests farm extensification. This new approach to farming systems could lead to reduction in diffuse nutrient concentrations in surface waters.

Under the perspective of the last CAP reform and further changes in land use, it is necessary to look at the proposals and their possible consequences. Issues like sustainability, organic farming, integrated farming and similar practices like no-till farming, which may become, due to the reform, again popular in UK must be considered. CAP reforms are intended on one hand to make European agriculture more competitive and orientated towards the market and on the other to remove all environmental negative incentives in the previous policy and encourage sustainable farming practices. The link between direct payments and production has been cut and intervention buying cancelled; which caused many of the problems with nutrients.

Payments are linked with achievements of environmental, food safety, animal welfare, health and occupational safety standards (Addiscott, 2005). Macgregor and Warren (2006) in their study on adopting sustainable farm management practices within a Nitrate Vulnerable Zone in Scotland found that many farmers are suspicious of the proposed de-coupling of direct payments from production. They suggest that current NVZ regulations will impact mostly on milk and intensive livestock producers and desired environmental benefits are unlikely to be realised because most farmers claim already to be managing nutrients within allowable limits.

2.4.3 Land Management Scenarios

Policy makers and scientists are interested in future land use changes and associated impacts on the environment, especially in respect of NATURA 2000, a large–scale European conservation and sustainability network project. Different directives can have great impact on environment, land use and rural populations. A large number of projects and studies have been undertaken to investigate future land use changes in Europe such as Advanced Ecosystems Analysis and Modelling (ATEAM) project (Metzger et al., 2006); EURURALIS (2003), ACCELLERATES (2001) and PRELUDE (Rounsevell et al., 2006-a).

The Foresight Programme (DTI, 1999; DTI, 2002) developed a methodology for construction of future agricultural and environmental scenarios for the UK, which considered long term futures and possible implications for UK industry and society (Morris, 2003). The Foresight Programme constructed four possible futures which are distinguished in terms of social values and governance (Berkout et al., 1998). The main drivers of future agricultural land use change in the EU and the UK are supply and demand due to population growth and economic prosperity, market intervention (agricultural policy), rural development policy, environmental policy, EU enlargement, resource competition (urbanisation, bioenergy crops), the role of the World Trade Organisation (WTO), social and environmental, issues, technology development, and climate change effects (agricultural productivity) (Rounsevell et al., 2006-b, Morris, 2003). These drivers are important as they generate different parameters (crop prices,

land use) which give the scenarios their particular distinguishing characteristics and meanings. Links between Foresight type scenarios and the scenarios for UK agriculture are presented in Table 2.5.

Table 2.5: Foresight Programme linked with the future Agricultural Policy Scenarios (Source: Morris, 2003)

'Foresight' Scenario	Agricultural Policy Scenario for UK	Intervention regime
	Baseline 'Business as usual' (CAP in 2002)	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.
Provincial Enterprise	Provincial 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, income and environment.

European studies on agricultural land use management changes for water quality improvement investigated different scenarios. Jessel and Jacobs (2005) in perspective of the WFD and CAP suggest that a scenario which includes 10 m wide buffer strips at the water edge is accepted by farmers in the Havel catchment, in Germany, and that with ongoing European agricultural reform, and single farm payments, alternation of arable land into grassland could become a more desired option. Rounsevell et al. (2006-b) developed land use changes scenarios, for Europe till 2080, and suggest a large reduction in agricultural areas for food production, partly compensated by increases in bioenergy production, forest land and areas protected for conservation and recreation. The same conclusion of a drastic decrease in land for agricultural purposes in the EU25 in the coming 30 years is shown in Van Meijl et al. (2006). Lenhart et al. (2003) suggest that deforestation and an increasing percent of intensively used grassland have the greatest effect on nitrogen discharges and water quality. However Eckhardt et al. (2003) suggest that in the case of change between forest and pasture and with regard to average stream flow (water quantity), the minimum proportion of catchment area on which land cover must change in order to obtain distinct model responses, amounts to about 25%.

2.5 MODELLING OF DIFFUSE POLLUTANTS

There are two methods for tracking the environmental fate of chemicals and assessing the effectiveness of diffuse source management techniques in preventing water quality deterioration: actual field monitoring and computer modelling (Shoemaker et al., 1994; Shirmohammadi and Knisel, 1994). Field monitoring has many limitations especially because of the variable natural elements (soil, geology, crops, climate), thus the collection of variable and large amounts of data for better understanding of natural catchment systems would be costly and time consuming. In this perspective computer models were developed, because they are viable alternatives in examining the environmental concentrations of diffuse pollution under different physiographic, climatic, and management scenarios (Shirmohammadi et al., 2000).

Computer technology development in the last two decades has had a great impact on study of the hydrologic system and has revolutionised water resources management. Computer models are in modern science used as a tool for better understanding and explanation of environment and its processes, and for predicting future events in a deterministic or probabilistic sense (Woolhiser and Brakensiek, 1982). They are helpful tools when the land use planner must decide whether a specified land management practice will meet soil or nutrient loss tolerance goals. Researchers can use the models to predict the effect of anticipated future land use changes on sediment and diffuse source nutrients delivery to the water body (Nearing, 2000) with estimates of certainty in their predictions. Nevertheless modelling is still just a tool for hypothetical predictions, and it is still not reliable if it is not based on sufficient data.

2.5.1 Modelling Approaches

Catchment models can be classified into three groups as empirical, physically based and conceptual models, and they can further be lumped or distributed (Whitehead, 2006). Empirical models (CREAMS, GLEAMS, MINDER, Export Coefficient) are often suitable only for modelling small catchments, with little variability, and when used in other catchments do not give satisfactory results. Physically based models (ANSWERS,

AGNSP, HSPF, IHDM, WEPP) are highly complex, and to get representative results much precise data is needed, which are often not available or even does not exist. Between these two modelling extremes is the most suitable group of conceptual distributed or semi-distributed models such as the Soil and Water Assessment Tool (SWAT). These models need less extensive data than others; the catchment can be subdivided into smaller units with their own characteristics and allow investigation of future behaviour in the river catchment both spatially and temporally.

Catchment scale hydrologic and diffuse source pollution models can be also divided as (Borah and Bera, 2003):

(1) Continuous simulation models useful for analyzing long-term effects of hydrological changes and catchment management practices, especially agricultural (SWAT, HSPF, ANSWERS-Continuous, AnnAGNPS);

(2) Single rainfall event models useful for analyzing severe actual or design single event storms and evaluating catchment practices, especially structural (AGNPS, ANSWERS, DWSM, KINEROS);

(3) Combined models with long-term and single-event simulation capabilities (CASC2D, MIKE SHE, PRMS)

For agricultural planning purposes, models that illustrate the relative advantages of different management practices may be more useful than models which produce absolute values (which may have little meaning to farmers and/or decision makers). In generating an overview of diffuse source pollution potentials, determination or prediction of long-term, annual average values may be more useful than forecasts of single, storm events (Jolankai et al., 1999).

Geographical Information System (GIS) have been applied to the analysis of water quality problems since the early 1980s (Logan et al., 1982) and their use has steadily increased. GIS offers tools for storage, processing, and visualization of spatially

distributed data (Shirmohammadi et al., 2000). Data in GIS are typically stored in two formats (vector or data) and georeferenced, and thus specified in relation to an earth centred coordinate system. Data in GIS are further characterized by their map scale which specifies their accuracy (Wolf and Brinker, 1994).

A common error made by model users is that they tend to consider the simulation results as true and absolute for unknown conditions. Output of a model may be affected by different errors: input errors, algorithm errors, incorrect or undue simplification of representing process in the model, size of the catchment etc. Diffuse source models try to represent the complexities of the natural environment with all its associated heterogeneities, thus they are never perfect (Shirmohammadi et al., 2000). However they can be used to investigate the occurrence of negative conditions before they develop or they can be helpful tool to investigate solutions for environmental problems like diffuse point source pollutants (Corwin, 1996).

2.5.2 Soil and Water Assessment Tool

The Soil and Water Assessment Tool (SWAT) is a semi-distributed, complex, conceptual, hydrologic, continuous time model specially developed to assist water resources managers in assessing the impact of management on water supplies and non-point source pollution in river catchments (Arnold et al., 1998). The core of the system was developed upon the 30 years of United States Department of Agriculture (USDA) modelling experience in the early 1990s. Three other models which contributed significantly to the development of SWAT were the Chemical, Runoff and Erosion from Agricultural Management Systems model (CREAMS) (Knisel, 1980), the Erosion-Productivity Impact Calculator (EPIC) (Williams et al. 1984), and the Groundwater Loading Effects of Agricultural Management Systems model (GLEAMS) (Leonard et al, 1987). SWAT was developed for use in ungauged catchments to predict impacts of land management on water, sediment and agricultural chemical yields (Arnold and Fohrer, 2005). It can be also used to predict impacts of climate changes. However, the model is semi-physically based and calibration is not possible on ungauged catchments. It is a spatially semi-distributed hydrological model, which means that the impact of

changes in spatially variable input parameters, such as land use, change can easily be modelled (Romanovicz *et* al., 2005). Major model components are hydrology, weather, soil temperature, plant growth, nutrients, pesticides and land management. The model is also capable of continuous simulation over long time periods, even for as many as a hundred years. SWAT operates on a daily time step and allows the catchment to be subdivided into natural sub-catchments containing a number of unique soil and land use combinations or Hydrologic Response Units (HRU-s) (Arnold et al., 1998; Shepard et al., 1999; Di Luzio et al., 2004). The water balance of each HRU is represented by four storage volumes: snow, soil profile (0-2 m), shallow aquifer (typically 2-20 m), and deep aquifer (>20 m). The soil profile can be subdivided in multiple layers. Soil water processes include infiltration, evaporation, plant uptake, lateral flow and percolation to lower layers and from there recharge to the shallow aquifer (Di Luzio et al., 2004).

A GIS - ArcView® 3.x and its extensions (Spatial analyst ™/ Dialog Designer ™) software interface is utilized to automate the assembly of the model input files from map layers and relational databases (Arnold et al., 1999). AVSWAT-X is the hydrological based system combined with GIS (ArcView® version 3.x) and SWAT 2005. This synergy (Arc View®GIS-SWAT) appears to be the key feature for an effective understanding and interpretation of complicated hydrologic processes connected with water quality assessment (Di Luzio et al., 2004). AVSWAT-X is further organized in a sequence of several tools grouped in eight modules: (1) Catchment delineation; (2) HRU Definition; (3) Weather station definition; (4) SWAT database; (5) Input parameterization, editing and scenario management; (6) Model execution; (7) Reading, mapping, charting results; (8) Calibration tool. AVSWAT also requires fundamental map inputs such as elevation (Digital elevation model - DEM), soil map, land use/cover and climate data.

The SWAT model is used in many countries all over the world. European Environmental Policy and implementation of the Water Framework Directive in 2000 demands tools like SWAT for integrative catchment management (Arnold and Fohrer, 2005). This is very flexible model and it can be applied to numerous different areas, under wide range of different environmental conditions.

2.5.3 SWAT Applications

SWAT is used throughout the world and has been accepted by the U.S. Environmental Protection Agency as one of their better assessment science integrating point and diffuse source (BASINS) models (Whittemore, 1998). As the model was developed in the USA many of the parameters used in SWAT available from existing databases are relevant to the USA but not to the UK. This overview of SWAT applications is focused on land management scenarios and diffuse source nutrients, as these are the main points of this study.

Studies showed the usefulness of the SWAT model for studying the effects of alternative management scenarios for pollution control from point and diffuse sources in catchments of different sizes. For estimating phosphorus loss from a lowland English catchment, SWAT was found to be the most suitable model of the 14 considered (Shepard et al., 1999) as it contains all physical characteristics of the ideal model such as a relevant scale and distributed nature. Santhi et al. (2001-a) studied impacts of management practices on dairy manure and WTW effluents on phosphorus loading and found SWAT to be adequate in predicting annual and monthly responses, and useful in analyzing management of dairy manure applications and waste water treatment plant effluents. Vache et al. (2002) studied the impact of three BMP scenarios on annual sediment and nitrate loadings and found SWAT to be a very useful tool in evaluating scenarios. SWAT was also found as very functional in Bracmort et al. (2006), for modelling long-term (~25 years) water quality impacts of structural BMP-s including consideration of the change in BMP effectiveness as their condition deteriorates. Santhi et al. (2006) found that the SWAT modelling approach is very useful for decision makers to assess the benefits of BMP-s on a catchment scale and suggested that the modelling approach could be extended to regional or national level and to other water quality projects and management strategies such as the European Union WFD.

Borah and Bera (2004) in a review of different models applications, suggest that SWAT is very suitable for predicting monthly and yearly flow volumes, sediment and nutrient loads; except for months with extreme storm events or hydrological conditions. The

model was also found useful for studying the impacts of climate changes on long-term water yields and impacts of management scenarios on long-term nutrient and sediment loads. Agricultural and agrochemical management simulations were found to be a unique strength of SWAT.

Chen and Mackay (2004) investigated how model structure and input data representation affect sediment predictions made using SWAT. They found conflicts, caused by the integration of HRUs and the Modified Universal Soil Loss Equation (MUSLE) and their conceptual incompatibility, making it difficult to determine the effect of different land use on soil erosion; they recommended that greater attention should be made to structuring the data inputs to avoid the spatial mismatches.

Jha et al. (2004) showed that catchment subdivision had very little effect on streamflow, but opposite results were found for sediment, nitrate and inorganic phosphorus. The optimal sub-catchment size, relative to the total drainage area was found to be around 3% for sediment, 2% for nitrate and 5% for inorganic phosphorus. Similar results that support the above statement were obtained in FitzHugh and Mackay (2000).

Barlund et al. (2006) tested SWAT-2000 model performance in the evaluation of management actions for the implementation of the Water Framework Directive in Finnish catchments and found that SWAT includes relevant management options that affect nutrient leaching. However, the description of some management options, like buffer strips, require modification to describe correctly the sediment reduction efficiency of buffer strips, as currently this is overestimated for local conditions.

Hydrological modelling of a small U.K. catchment using SWAT-2000 was performed by Kannan et al. (2007) and as a result new modifications were introduced to SWAT. It was found that proper modelling of water balance components such as crop growth and evapotranspiration is crucial for correct representation of flow pathways, and that the SWAT model can be reliably used to model stream flow in UK conditions.

3 METHODOLOGY

3.1 STUDY AREA AND AVAILABLE DATA

3.1.1 Topography and Geography

The River Axe catchment is situated in the south-west of the United Kingdom in the counties of East Devon, West Dorset and South Somerset and is a part of the Atlantic biogeographic region. The total catchment extends over an area of approximately 400 km^2 (Figure 3.1). As this is a coastal catchment, altitude varies between 0 m and a maximum of 316 m above sea level (CEH, 2006). The three main towns in the catchment are Chard (population 12,000); Axminster (population 6,000) and Seaton (population 6,500). Other settlements are mostly small dispersed villages and hamlets with population less than 500 (Cycleau, 2004).

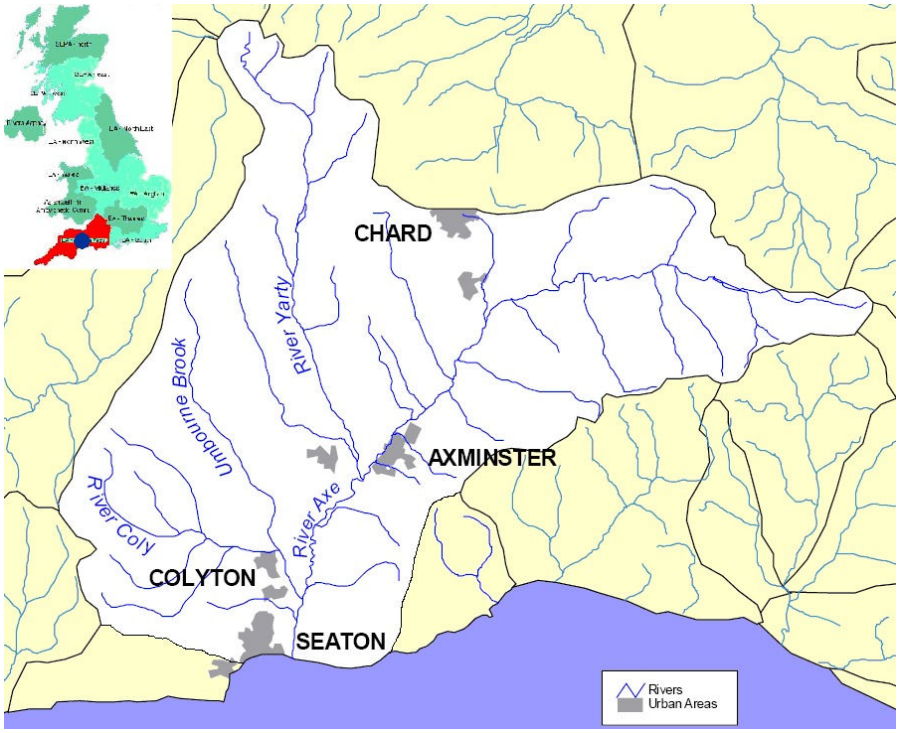


Figure 3.1: The river Axe catchment area showing major tributaries and urban areas (Source: Ash et al., 2005)

The main characteristics of the catchment are very steep slopes between tributary valley ridges (River Yarty, Umborne Brook and Corry Brook) and flood plains along the river Axe main channel. The Blackdown Hills are situated in the west and north-west part of the catchment with the highest altitudes in the catchment (316 m) and divide the river Axe and Otter valleys (Figure 3.2). The main river valley of the River Axe is relatively wide with a flat valley floor across which the river meanders (Environment Agency, 2004-b).

Figure 3.2: View from the eastern part of the river Axe catchment towards the west (Blackdown Hills).

The valley is shaped in a flat letter U (Figure 3.3), as a result of glacier melt at the end of the last glaciation period. The main features in the south part of a catchment are coastal chalk plateaux and cliffs. The valley of the River Axe progressively widens in a downstream direction as successive tributary streams join the main river. Below the tidal limit the Axe joins the sea through an estuary, which is narrower that it was in the past. This area is of ecological importance as it contains mudflats and areas of salt (Environment Agency, 2004-b).

Figure 3.3: View on the River Axe estuary and coastal chalk plateaux.

3.1.2 Geology and Soils

The catchment geology (Figure 3.4) primarily consists of Mesozoic (Triassic, Jurassic and Cretaceous) strata locally overlain by Tertiary (Eocene) deposits (Environment Agency, 2004-b). Generally it is a catchment of moderate relief, draining Chalk and Greensand headwaters. The western part of the river Axe is underlain by Triassic Mercia mudstone. The Upper Axe is underlain by the Blue Lias formation (Jurassic). These are overlain by the Upper Greensand (Cretaceous) and also by chalk (on isolated hilltops mainly to the east). As a product of the erosion of these formations, the lower part of the Axe meanders through deep sandy, muddy alluvium within a well-developed floodplain (Cycleau, 2004; JNCC, 2006).

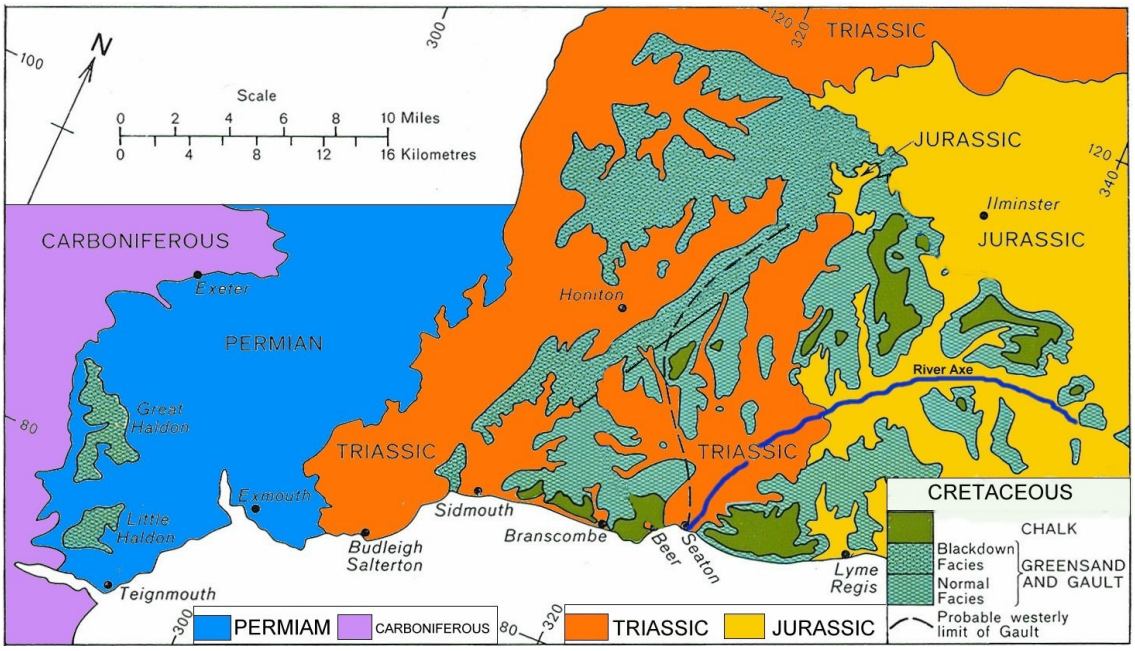


Figure 3.4: Geology of eastern Devon and the river Axe catchment (Duff and Smith, 1992)

The soils of the catchment reflect its geology. A variety of brown earth soils have developed in the upper catchment on the Greensand and Clays, these include argillic and stagnogleyic varieties. In areas adjacent to the river and stream channels the gley types dominate. In the lower reaches of the catchment the poorly draining floodplains are dominated by palaeo-alluvium gley soils. The lower ground bordering the slopes also contains a range of soils including gleyic argillic brown earth, humic gleys (organic rich) and argillic palaeosols (Sweet, 2004; Environment Agency-b, 2004). The national soil map (NATMAP, 1999) gives us more precise information on soils series and their characteristics (Figure 3.5, Appendix A). Major fractions of the catchment are represented by Whimple 3 (572f) (17%), Batcombe (582a) (15%), Bearsted 2 (541B) (11%), Wickham 2 (711f) (9.7%) and Charity 1 (571l) (9.3%) soil associations. The majority of these soils are loamy or silty over clay with slowly permeable sub-soils and are thus very sensitive to inappropriate management of land.

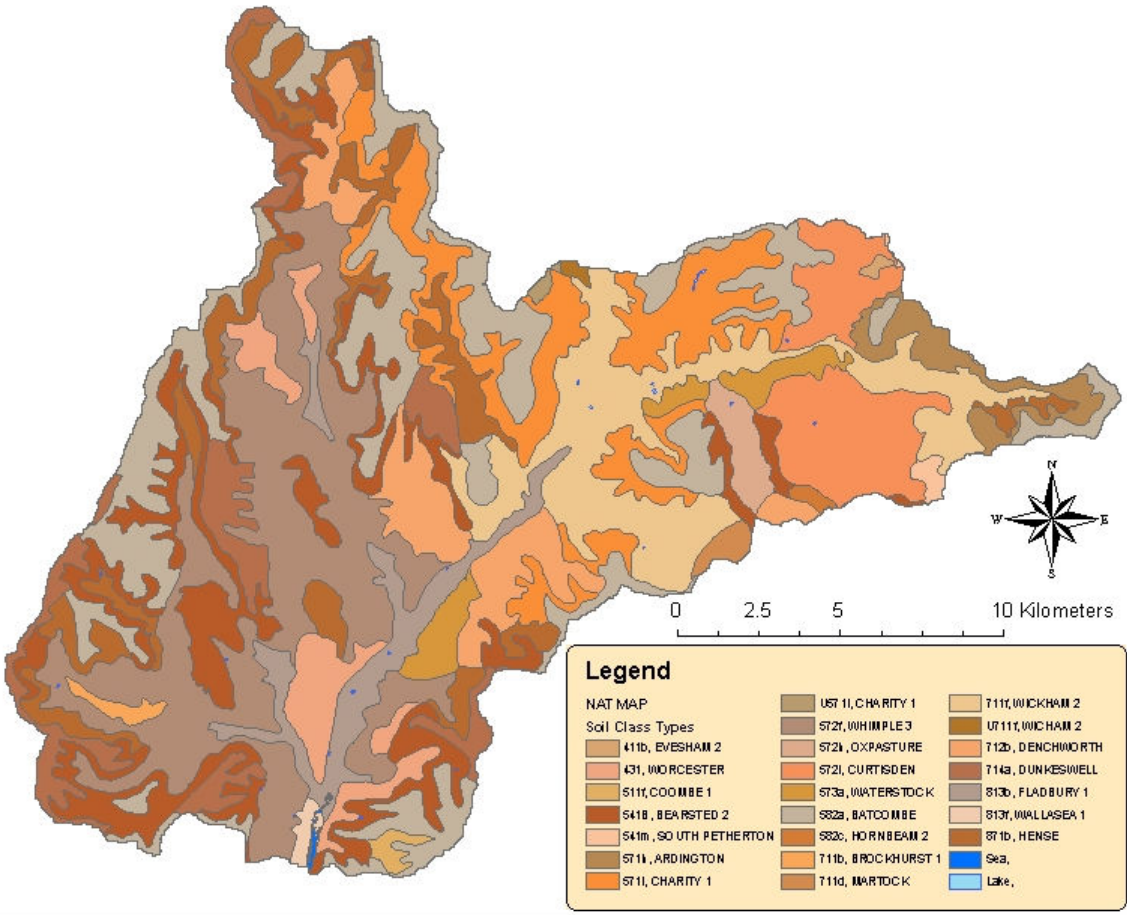


Figure 3.5: Soil map of the river Axe catchment (SSLRC, 1983).

3.1.3 Hydrology and Hydrogeology

The River Axe rises in the Somerset Hills at elevation 175 m above sea level, just south of Cheddington. Initially the river flows in a westerly direction before turning south and flowing to the tidal limit, a total distance of 45 km until it enters the sea at Seaton (JNCC, 2006; Environment Agency, 2004-b). The broad Axe valley runs through an unbroken agricultural landscape, joining with numerous tributaries like the River Yarty, Corry Brook and Umborne Brook, all of which originate on the steep slopes of the Blackdown Hills (Figure 3.6). The River Coly discharges into the Axe estuary just south of Colyton (Ash et al. 2005). The hydrology of the area is characterised by a fast response to rainfall, with rapid runoff typical for low permeability catchments. This is in

part responsible for the natural regime of erosion and for the re-deposition of sediment (Cycleau, 2004).

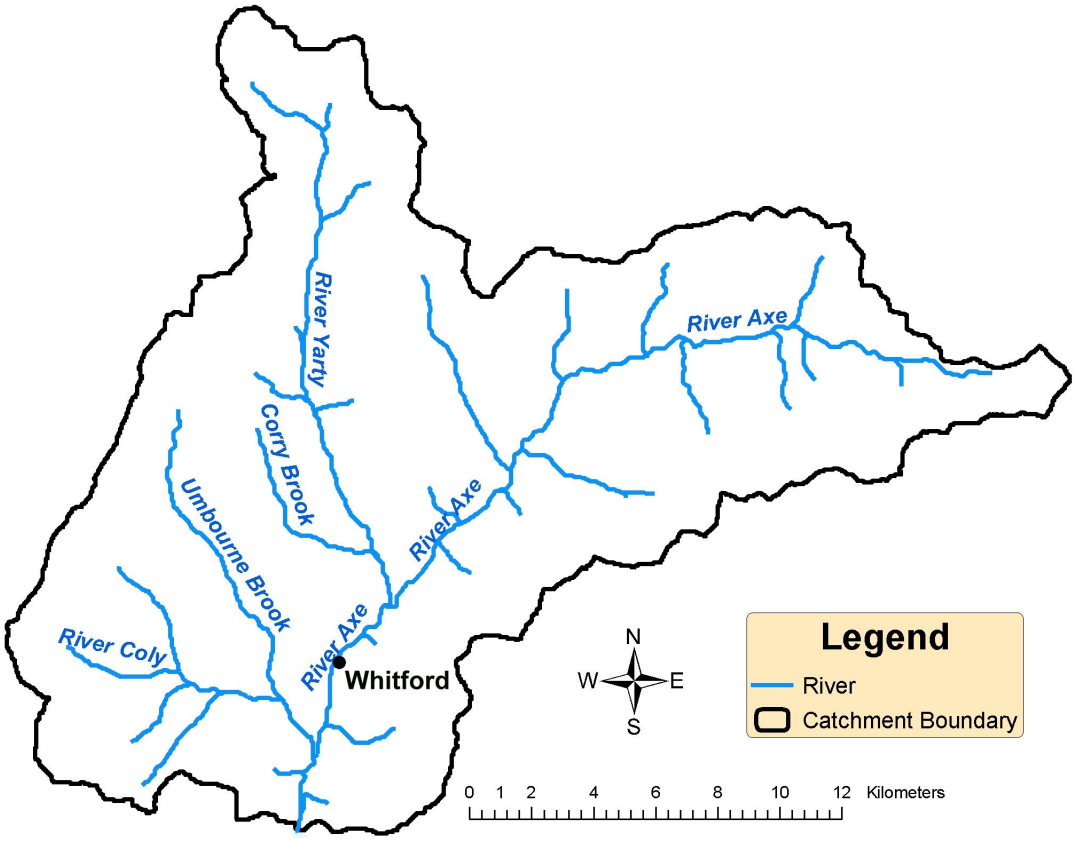


Figure 3.6: River network of the Axe catchment.

River flow has been recorded at Whitford Bridge station since 1964; this shows a mean daily flow of 5.11 $m^3 s^{-1}$. Flow at Whitford equals or exceeds 1.241 $m^3 s^{-1}$ 95% of the time, and the flow with probability of 10 % exceedance (Q10) is 11.05 $m^3 s^{-1}$ (CEH, 2006). The maximum recorded flow of 244 $m^3 s^{-1}$ which was recorded on 27th December 1979 (Environment Agency, 2004-b). Hydrological records indicate that base flow is high, reflecting the soils and subsoil and the presence of large aquifers within the catchment (Environment Agency, 2003). Groundwater makes a significant contribution to base flow and maintains river flows during dry weather. River flow is affected by different factors. It is reduced by public water supply, industrial and agricultural abstraction and evapotranspiration; increased by precipitation and effluent returns (sewage and water treatment works) and influenced by groundwater abstraction and recharge. Figures 3.7, 3.8 and 3.9 show the river Axe daily mean flow, monthly

mean flow, and yearly mean daily flow, respectively for the period from 1986 to 2005. The mean daily flow is highly variable with peak flows in winter months (December, January and February) while low flows occur during summer months (July, August). The average mean monthly flow has shown a slight rise over the period 1986-2005. This appears to be mainly the result of higher flows during winter months (Figure 3.7).

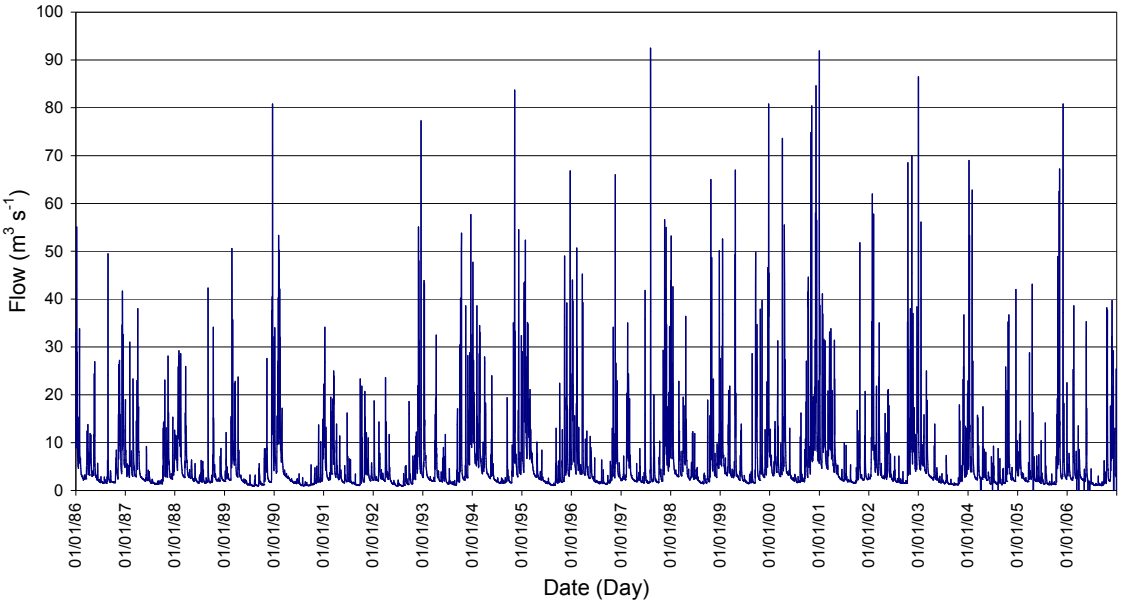


Figure 3.7: The River Axe daily mean flow at Whitford gauging station from January 1986 to December 2005.

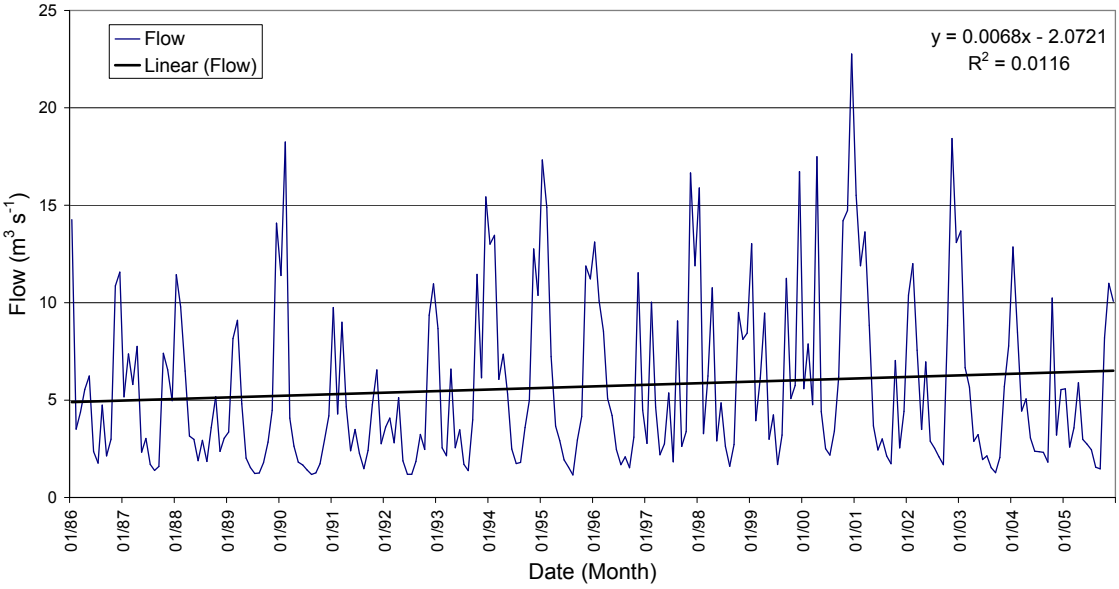


Figure 3.8: The River Axe mean monthly flow at Whitford gauging station from January 1986 to December 2005.

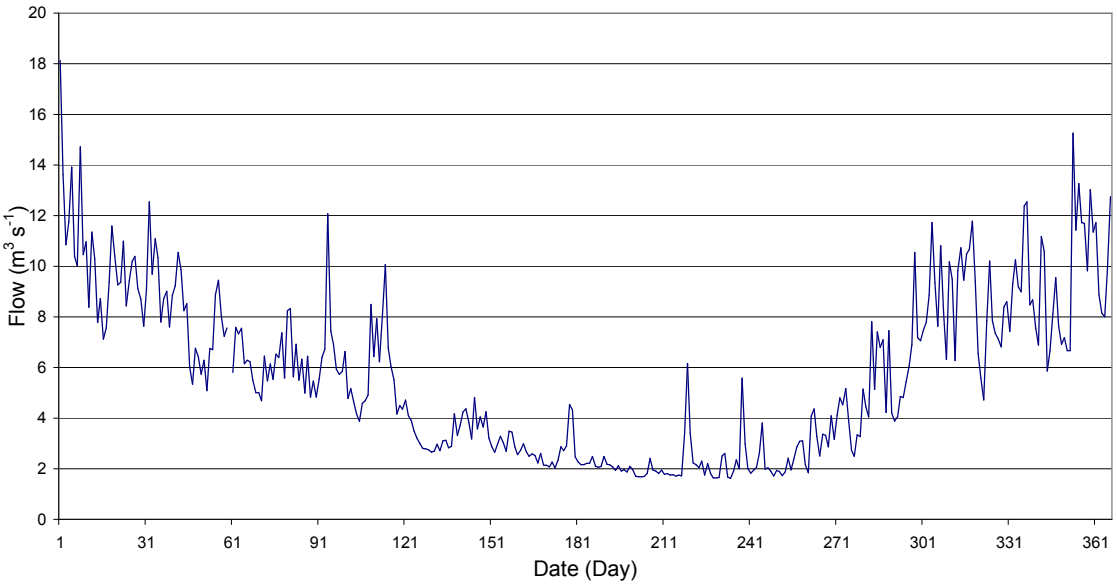
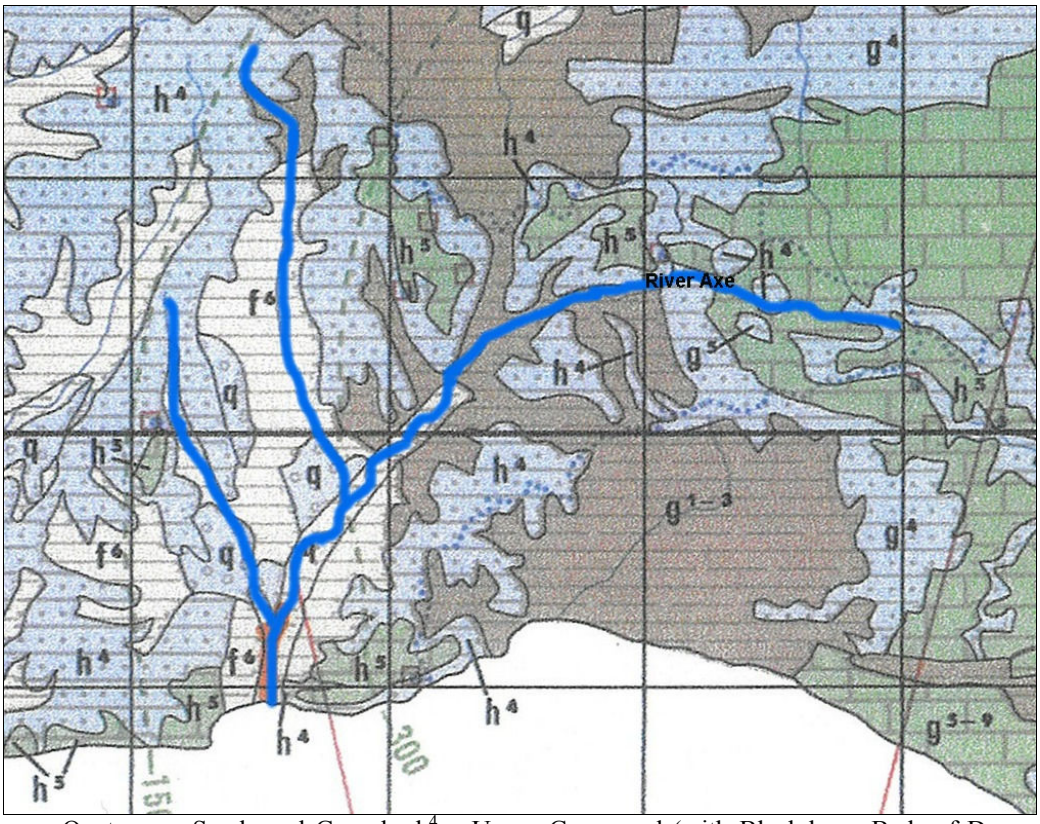


Figure 3.9: The River Axe yearly mean daily flow at Whitford gauging station for the period from January 1986 to December 2005.

Hydrogeology of the study area is diverse. The hydrogeology of the catchment is classified in three different types. The higher north-east parts of the area are highly fissured and permeable; other higher parts have moderate intergranular permeability, the lower parts of the valleys have very low permeability.

The main aquifers in the catchment consist of the Cretaceous Greensand and Chalk strata (Figure 3.10). In addition the alluvium contained within the river valleys provides minor aquifers where high ground water storage and flow provide recharge during low flow conditions (Environment Agency, 2004-b). Locally important aquifers, in which flow is dominantly intergranular like Upper Greensands, Quaternary Sands and Gravels, and Upper Lias, can be found all over the catchment.



Key: q – Quaternary Sands and Gravels; h^4 – Upper Greensand (with Blackdown Beds of Devon and Dorest; h^5 – Chalk; f^6 – Triassic: Keuper Marl; g^{1-3} – Lias, g^4 – Upper Lias: Bridford, Midford, Yeovil and Cotteswold Sands; g^{5-9} – Cornbrash, Great Oolit, Fullers Earth and Inferior Oolite of South England

Figure 3.10: Hydrogeological map of the river Axe catchment (IGS, 1977).

The catchment is not at risk from over abstraction. Over 80% of the abstracted water is from surface water of which 34% is used by industry, 27% provides public water supply and 22% is consumed by agriculture (Environment Agency, 2003). Water is also abstracted for private water supplies.

3.1.4 Climate

The area experiences a temperate Atlantic climate. Average annual rainfall ranges from 820 mm in the lowland areas of the Axe estuary, to over 1100 mm in the north-west, upper catchment in the Blackdown Hills and North Dorset Down altitudes. Average annual precipitation for the study area is 1063 mm. Figure 3.11 shows 20 years annual average precipitation for the period of 1986-2005 for 9 rainfall gauging stations used in the SWAT model.

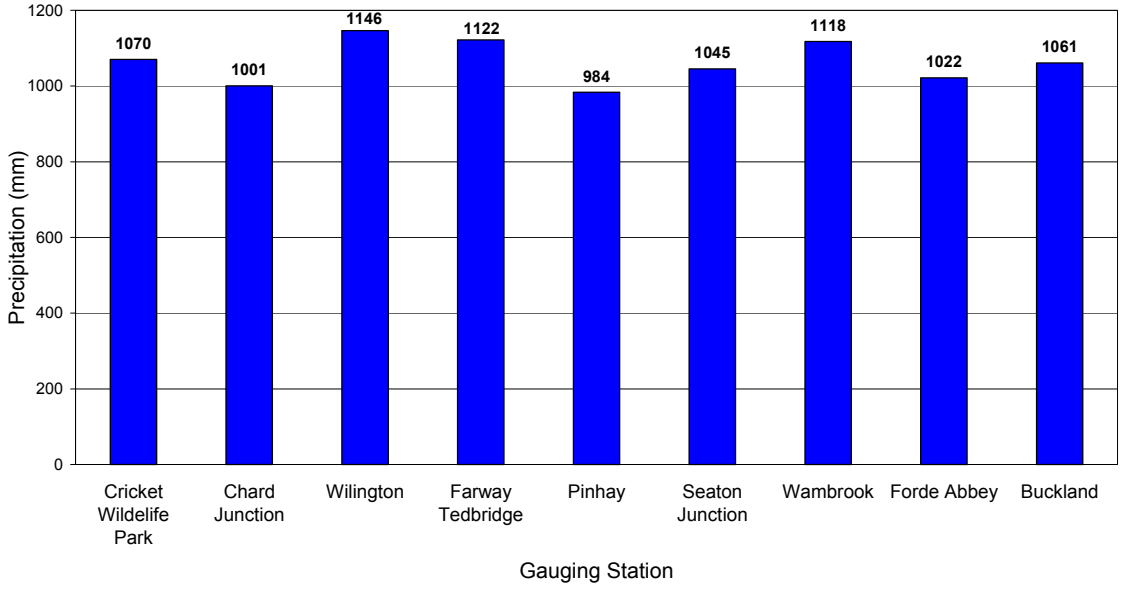


Figure 3.11: Mean annual precipitation for nine rainfall gauge stations in the study area for the period from 1986 to 2005.

Figure 3.12 shows annual precipitation values recorded at Cricket Wildlife Park in the north of the catchment. Data shows the highest recorded precipitation in 2000 (1447 mm), 2002 (1254 mm) and 1999 (1364 mm). The lowest precipitation was recorded in 2003 (844 mm), closely followed by 1991 (845 mm), 1992 (864 mm), and 1988 and 1990 (both 871 mm). Annual recorded precipitation (mm) in other parts of the catchment follows the same pattern and can be observed in Appendix B.

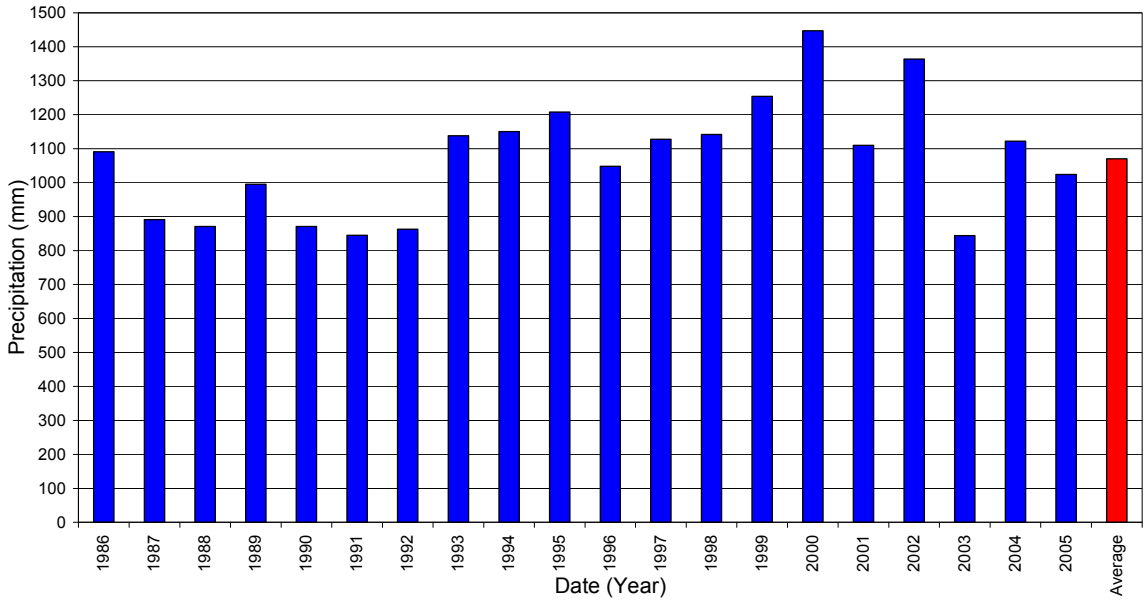


Figure 3.12: Total annual precipitation and 20 years annual average for Cricket Wildlife Park gauge station for the period of 1986 to 2005 (BADC, 2007)

The highest monthly precipitation occurs in autumn and winter months (October - January) and the lowest precipitation occurs in summer months (May - August) (Figure 3.13).

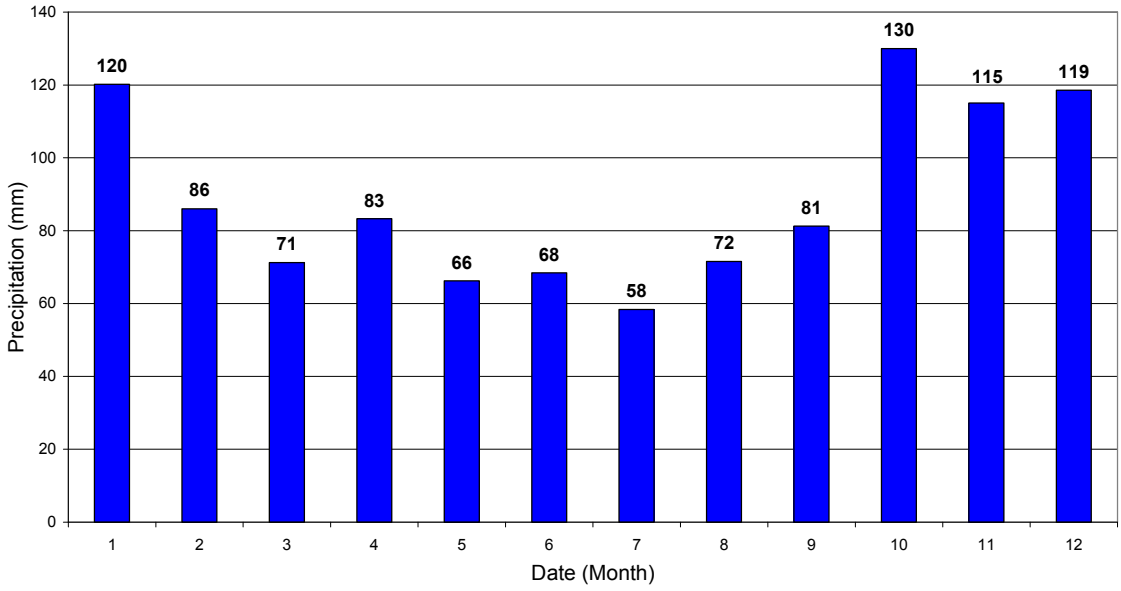


Figure 3.13: Twenty years monthly average precipitation for Cricket Wildlife Park gauge station for the period from 1986 to 2005 (BADC, 2007)

As there is no currently active temperature gauging station in the study area, data for 20 years from 1986 to 2005, from neighbouring stations were obtained. The coldest months at Hemyock (north-west) temperature station is February with average monthly temperature 4.9°C and the warmest months is August with 16.1°C (Figure 3.14). The coldest months at Sidmouth (south-west) is also February with temperature 6.1°C and the warmest month is August 17.0°C (Figure 3.15).

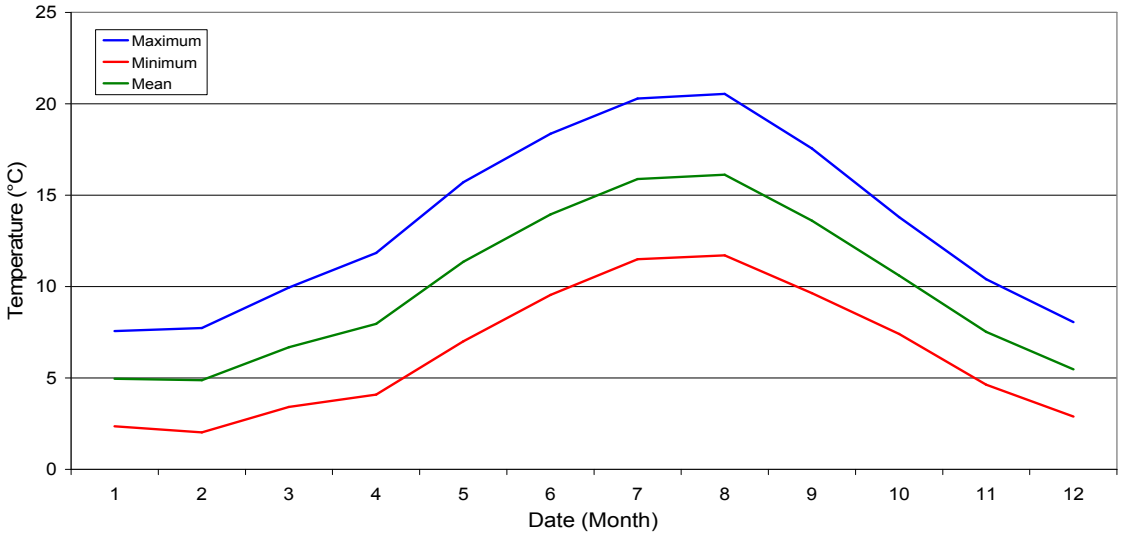


Figure 3.14: Monthly average (maximum, minimum, and mean) temperature (°C) for Hemyock for the period 1986 to 2005.

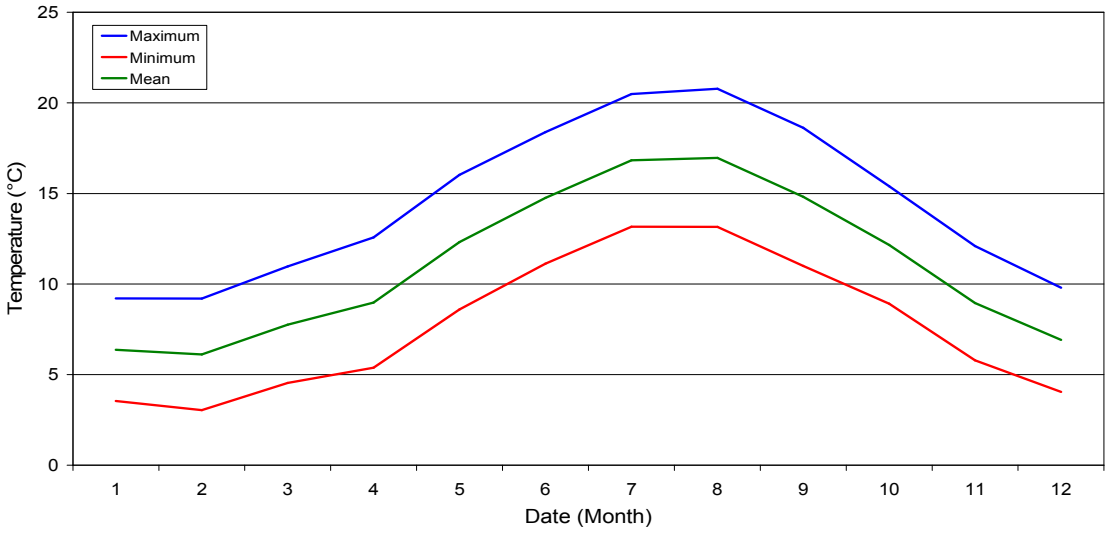


Figure 3.15: Monthly average (maximum, minimum, and mean) temperature (°C) for Sidmouth for the period 1986 to 2005.

As can be seen from Figure 3.16 for Hemyock and Figure 3.17 for Sidmouth, the daily mean temperature has been constantly rising over study period. High summer temperatures are becoming more frequent, while cold winters are becoming increasingly rare.

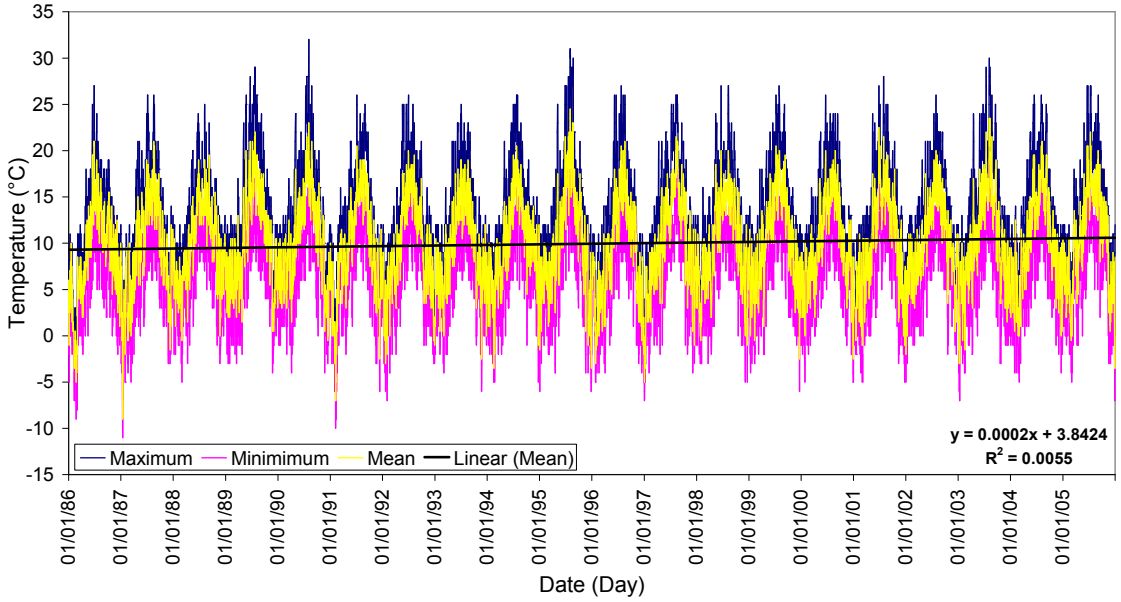


Figure 3.16: Maximum, minimum, and mean temperature (°C) for Hemyock for the period from 1986 to 2005.

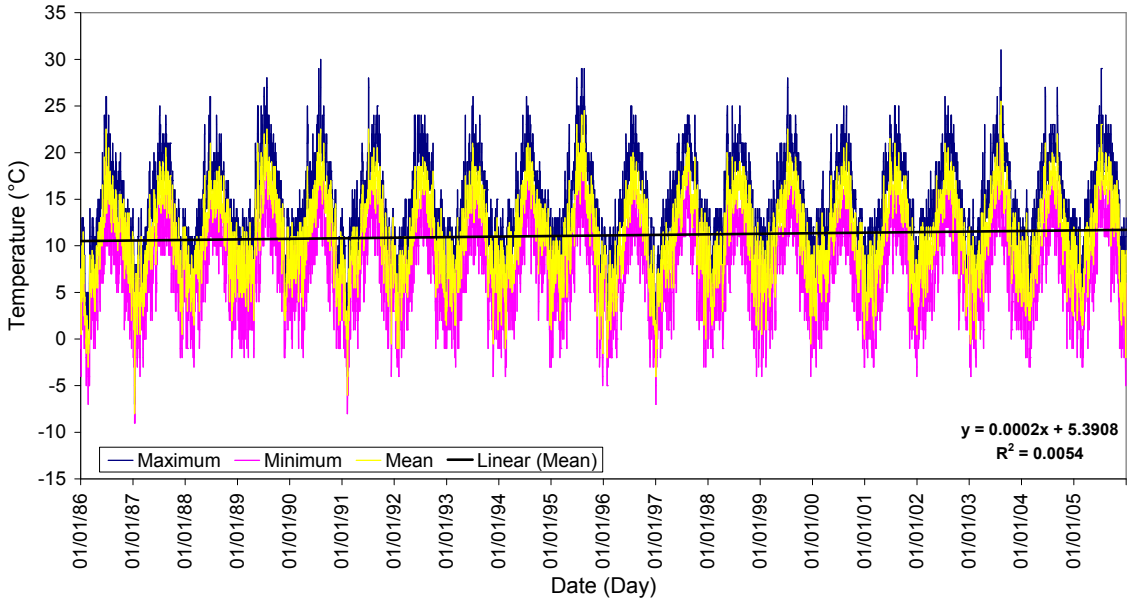


Figure 3.17: Maximum, minimum, and mean temperature (°C) for Sidmouth for the period from 1986 to 2005.

3.1.5 Land Use

A high percentage of agricultural land, approximately 94% is a characteristic of the River Axe. The CORINE 2000 (CLC, 2000) land use map shows (Figure 3.18) that the majority of the catchment area is in use as grassland – pastures (CORINE class 231) with 73%, followed by 21% of different types of arable land (CORINE class 211, 242, 243) and 4% of forest (CORINE class 311, 312, 324). The spatial distribution of the different agricultural use is generally highly mixed; therefore it is difficult to determine areas of uniform land use. The upper valleys of the catchment are wooded; this gives way to a landscape dominated by small fields and hedges. The lower catchment is dominated by an agricultural landscape of pasture land, mainly used for dairy farming, with arable farming primarily for maize cultivation (Environment Agency, 2004-b).

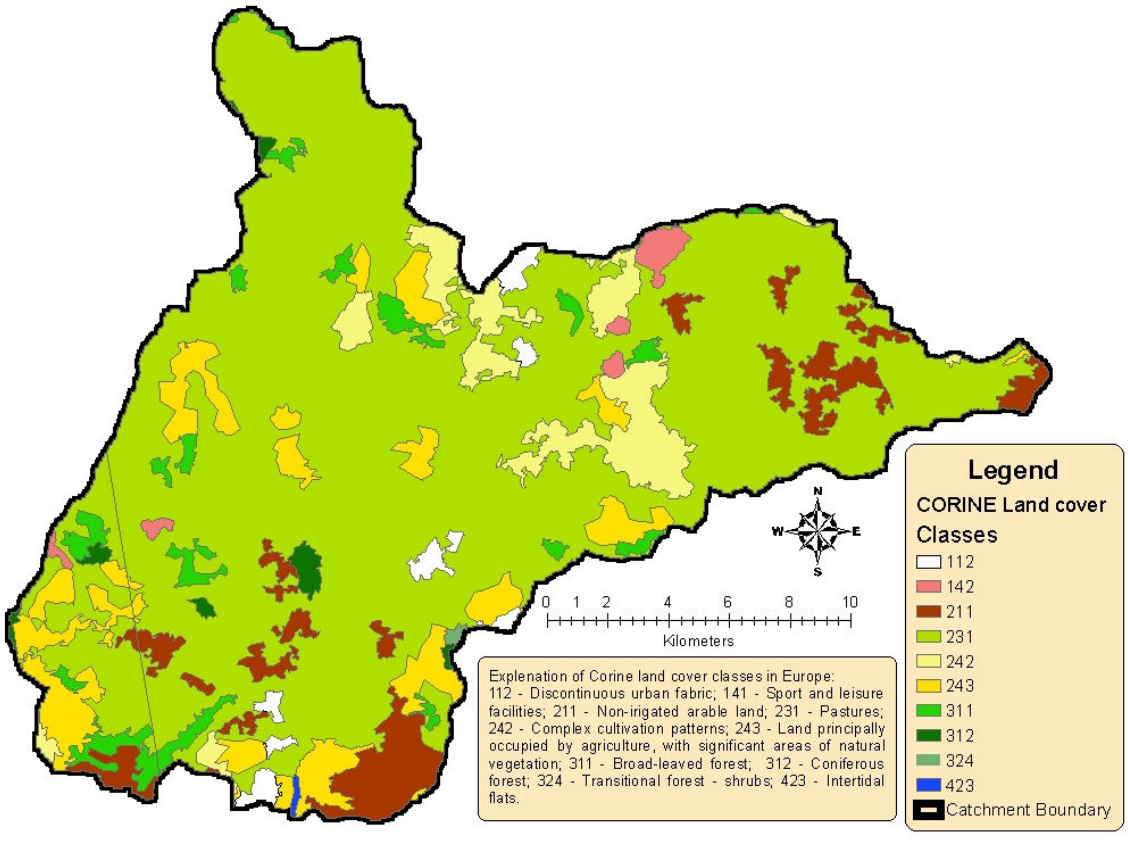


Figure 3.18: Land use in the river Axe catchment (CLC, 2000)

Livestock production, which is based on dairy cows and sheep, is the major form of agricultural management in this catchment. This is clearly visible from the structure of

land use where grassland for grazing and grass silage predominates (73%). Arable areas are mainly seeded with maize, winter wheat and winter barley as fodder for dairy cow production demands (EDL, 2006). From Figure 3.19 it can be seen that the area of maize has been constantly increasing through the years mainly at the cost of winter cereals. A high fraction of arable land (51%) is designated to intensive temporary grassland with cut grass silage management. Areas (ha) and fractions (%) covered by individual crops are shown on Figure 3.19 and Figure 3.20 respectively.

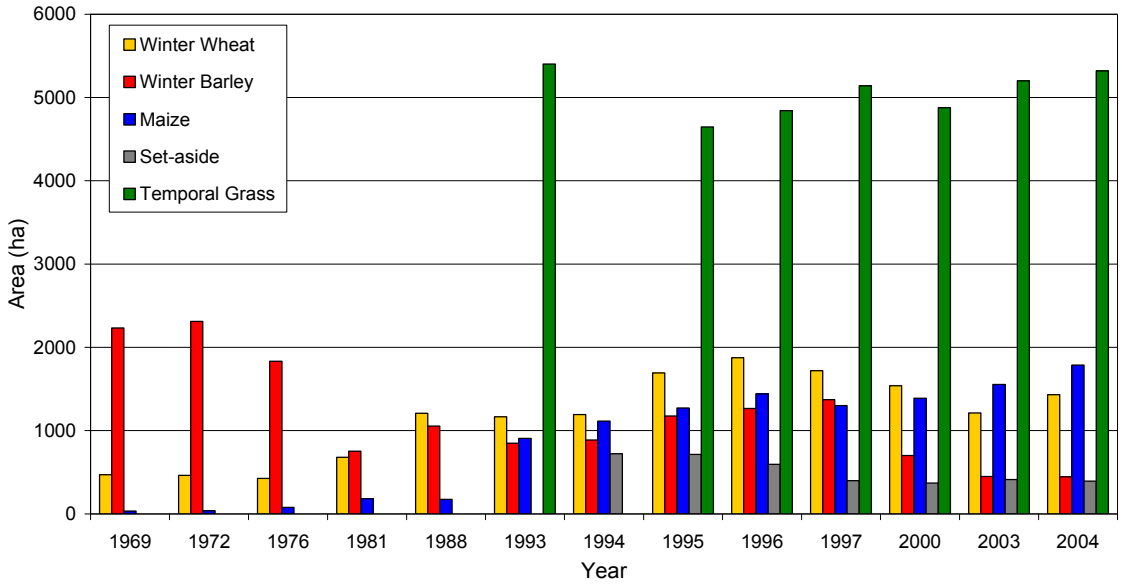


Figure 3.19: Area (ha) covered by different crops between years 1969 and 2004 (EDL, 2006)

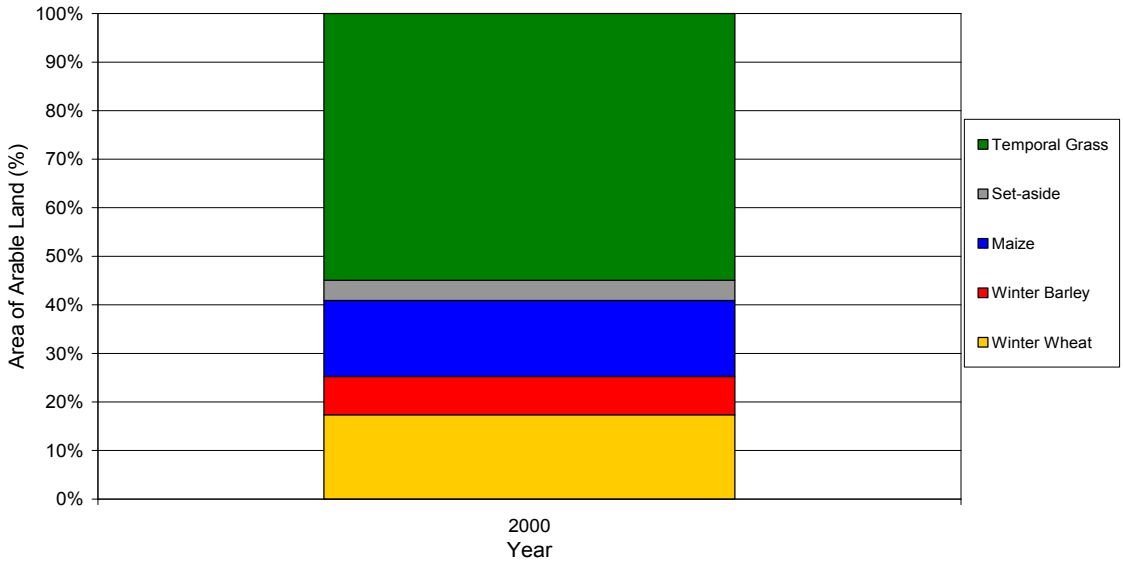


Figure 3.20: Fraction of individual crops in study area arable land for year 2000 (EDL, 2006)

3.1.6 Livestock

Dairy cow and sheep livestock management are the main types of farm production in the study area. Figure 3.21 shows that the number of dairy cows has fallen from the beginning of the 1990s, from more than 22,000 to less than 19,000 (EDL, 2006). This negative trend in number of dairy cows is a consequence of farmer retirements without successors and low prices of milk (CAP) and disease (foot and mouth). However, because of the low prices of milk the remaining farmers have had to raise efficiency and the consequence can be seen from the EDL (2006) data for arable land especially from the area of maize (Figure 3.19). In the year 1993 maize covered approximately 900 ha and in 2004 this had increased to over 1,780 ha. This clearly shows, that lower numbers of animals does not directly mean lower intensity of farming; it can mean even higher intensity and more destructive farming practices.

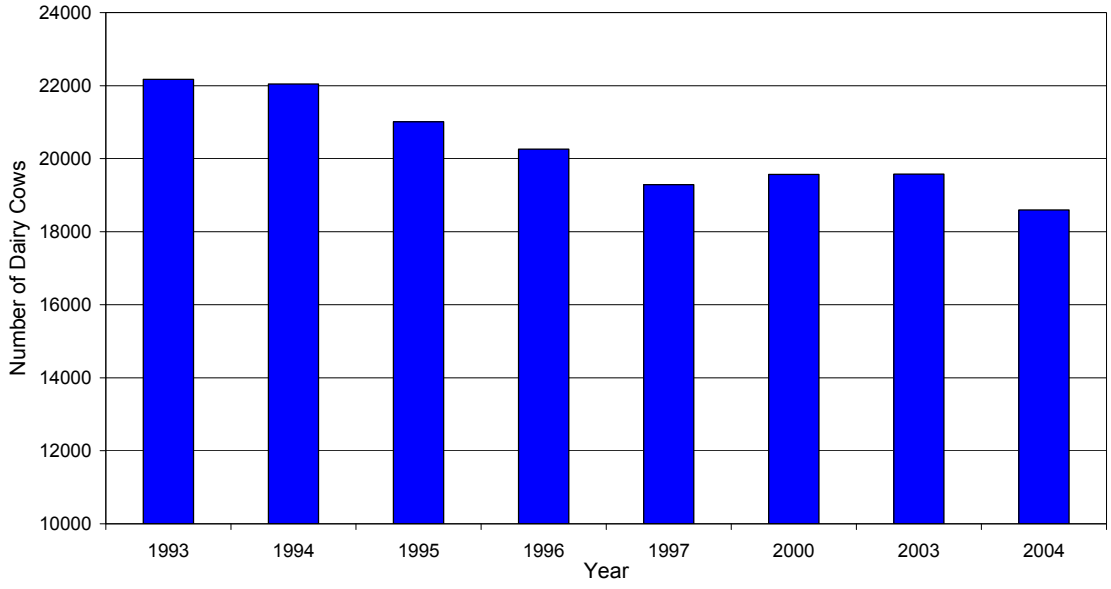


Figure 3.21: Number of dairy cows in the River Axe catchment between 1993 and 2004 (EDL, 2006).

Overall numbers of sheep in the study area were rising over the period from 1969 (26,205) till 1988 (51,731). The number of sheep is presently stable at approximately 50,000 in the study area (Figure 3.22).

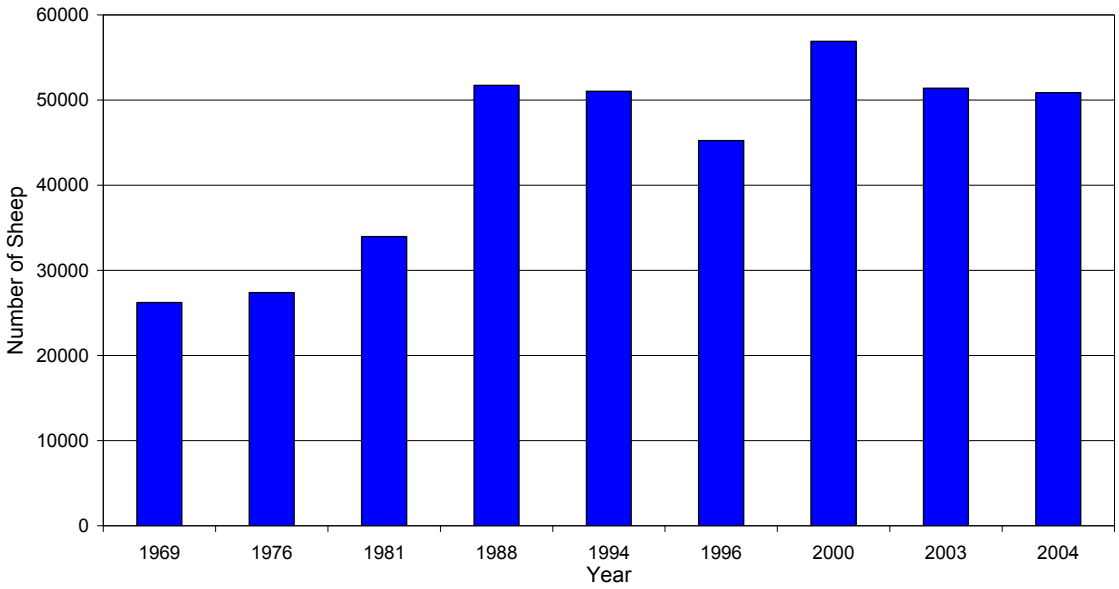


Figure 3.22: Number of sheep in the River Axe catchment between 1969 and 2004 (EDL, 2006)

3.1.7 Water Quality and Sources of Pollution

Sources of pollution in the River Axe catchment are very dispersed across the area, due to numerous sewage treatment works (STW), diverse land use management techniques and livestock production. Therefore nutrients can come from both point and non-point (diffuse) sources.

In year 2000 the Axe catchment was graded good or fairly good for chemical quality and good or very good for biological quality, using the national classification scheme, General Quality Assessment (GQA) (Environment Agency, 2004-b). However, Temple Brook was graded poor for chemical quality. Water quality has shown deterioration in recent years. By 2002, the majority of the Axe and Yarty were failing to meet River Quality Objectives (RQO). As the Environment Agency (2003) report suggests deterioration has been attributed to increased biochemical oxygen demand (BOD) and ammonia occurring mainly as a result of agricultural pollution with additional inputs form sewage treatment works (STW) discharges and storm sewer overflows. High levels of phosphorus have resulted in eutrophication, which is most severe in the lower catchment. Nitrate levels are moderate in the upper catchment, and high in the ground

waters of the lower catchment. High suspended sediment levels (>1000mg l^{-1}) have been recorded in the catchment at the Whitford Bridge. The mean level of suspended solids at Whitford Bridge in 2002 was 57.7 mg l^{-1}, well in excess of the Environmental Quality Standard (EQS) of 25 mg l^{-1} (Environment Agency, 2003). Table 3.1 represents the EA nitrate and phosphorus classification for surface waters.

Table 3.1: Environment Agency EQS nitrate nitrogen and phosphorus classification scheme (MacDonald, 2006; Environment Agency, 2006-b).

Quality Grade	Nutrient Grade Limits			
	Nitrate nitrogen (mg NO_3-N l^{-1})		Phosphorus (mg PO_4^{3-} l^{-1})	
1	<1.1	Very low	<0.02	Very low
2	>1.1 to 2.3	Low	>0.02 to 0.06	Low
3	>2.3 to 4.5	Moderately low	>0.06 to 0.1	Moderate
4	>4.5 to 6.8	Moderate	>0.1 to 0.2	High
5	>6.8 to 9.1	High	>0.2 to 1	Very high
6	>9.1	Very high	>1	Excessively high

Water quality grade at Axe Bridge for nitrate-nitrogen in the River Axe is between 3 and 4 (Figure 3.23) with average annual concentrations between 4 and 5 mg NO_3-N l^{-1}. By the EA Quality Standards nitrate scale these values do not represent a large threat to the river. However average water quality grade is in the moderate state and data for the last three years shows a small growth in concentrations, therefore should be more emphasis on nitrates too (Figure 3.24).

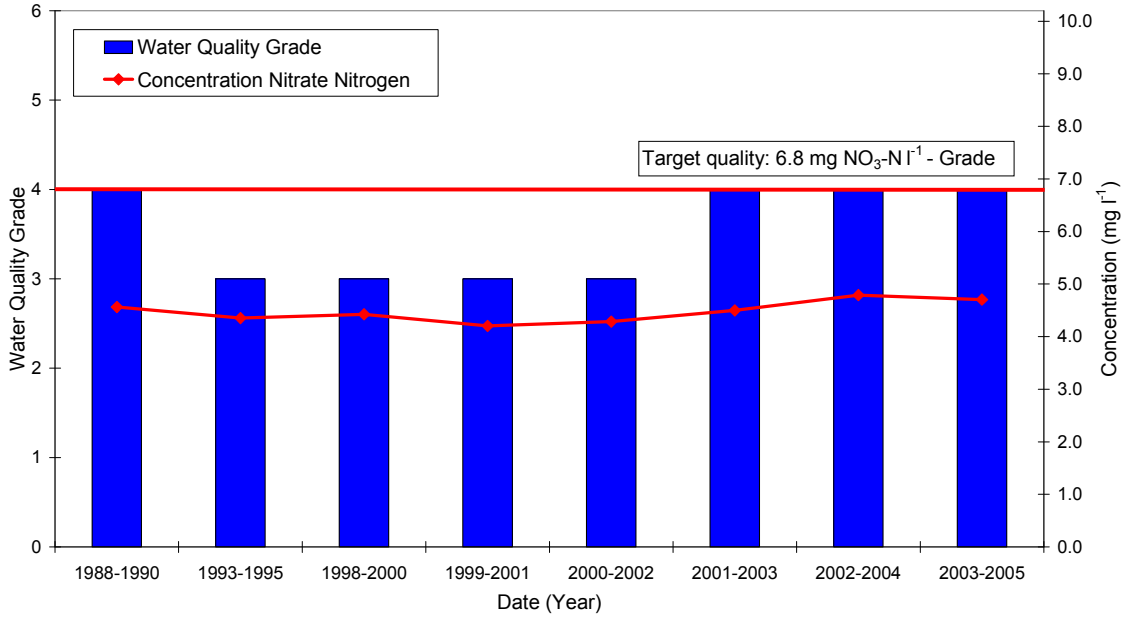


Figure 3.23: Water quality grade for nitrate nitrogen (NO_3-N) at the River Axe tidal limit (Axe Bridge).

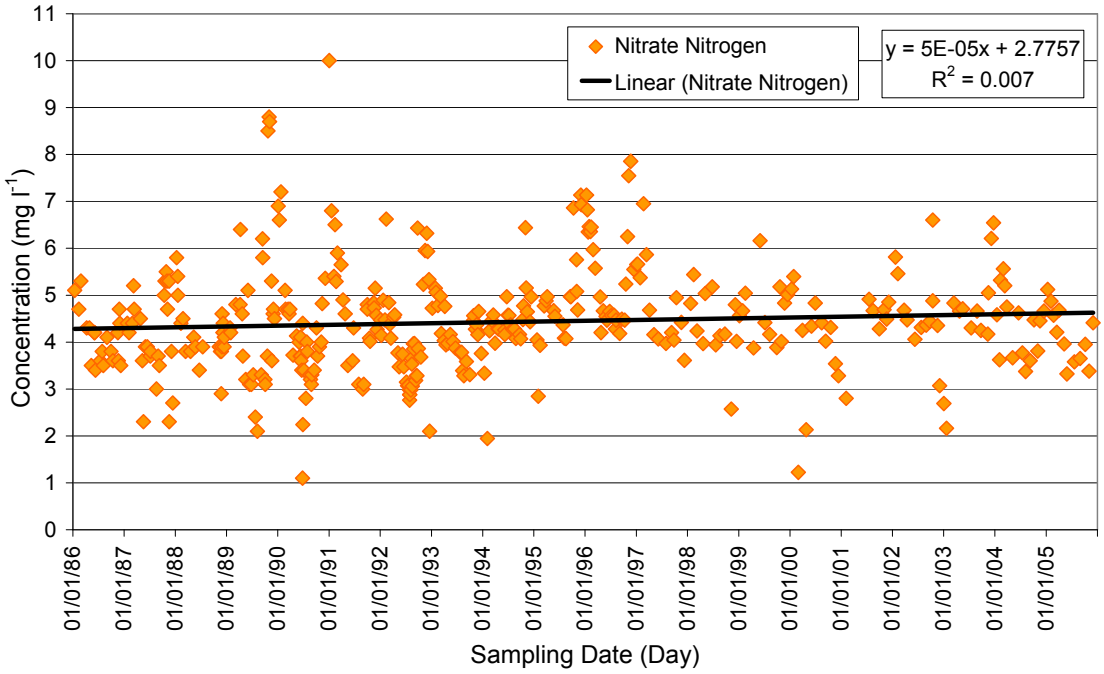


Figure 3.24: Measured concentration of nitrate nitrogen (NO_3-N) at Whitford gauging station for the period from 1986 to 2006.

The ortho-phosphorus water quality grade at Axe Bridge is 5 (Figure 3.25) with average annual concentration of 0.23 mg PO_4^{3-} l^{-1} for the period 2003 - 2005, exceeding the target grade 2 (0.06 mg l^{-1}). Phosphorus is, according to the EA guideline standards, the biggest threat for the Axe ecosystem. EA research (Ash et al., 2005) has shown that diffuse sources of pollution contribute to the Axe annual mean phosphorus concentration approximately 53.38% of total annual load.

Walling et al. (2000) carried out a study on river flood plains as phosphorus sinks, on 20 British rivers, which showed surprising results. The total phosphorous content of deposited sediment in the river Axe flood plains has dramatically increased by 170% over the period from 1950 to 1992. They suggest that these increases are connected to the more intensive agriculture in the river Axe catchment, which includes a substantial proportion of arable cultivation and significant increases in fertiliser application over the past 40 years.

Point sources of nutrients from sewage treatment works (STW) and industrial discharges are also big contributors to total nutrient loads in the rivers. There are over

400 licensed discharges in the Axe catchment. Point source load contributions of phosphorus were estimated at approximately 30 tonnes per year, or 47.6 % of total phosphorus load in the River Axe (Ash et al., 2005). The three most important dischargers are at the St Ivel Creamery Ltd. and STWs at Axminster and Seaton. The Environment Agency (Ash et al., 2005) has performed a SIMCAT mathematical modelling exercise for the Axe catchment to predict water quality and phosphorus concentrations. The results showed (Figure 3.26) numerous points where phosphorus levels exceed target values of 60 µg l^{-1} (0.06 mg l^{-1}), due to the point sources. High concentrations of phosphorus in river water represent a serious threat to biodiversity in the catchment. However water quality has improved since 2002 as phosphorus stripping was introduced to the main sewage treatment works with population over 10,000 (discharges now achieve less than 2 mg PO_4^{3-} l^{-1}) and to the main industrial contributor of phosphorus, St Ivel Creamery Ltd. The results of the introduction of phosphorus stripping can be observed in Figure 3.27.

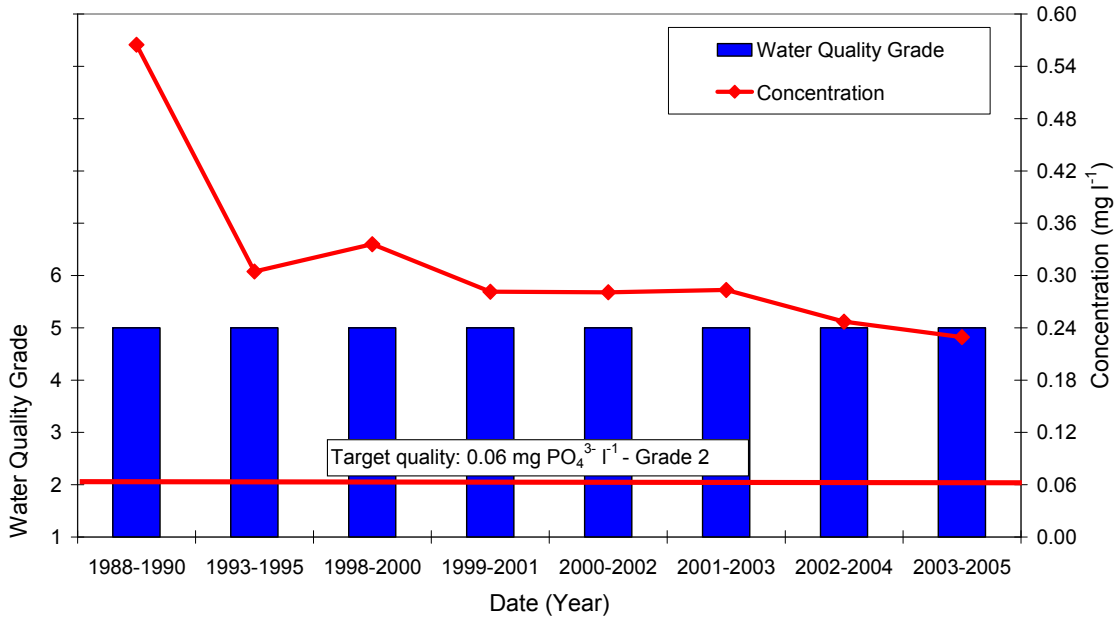


Figure 3.25: Water quality grade for ortho-phosphorus at the River Axe tidal limit (Axe Bridge).

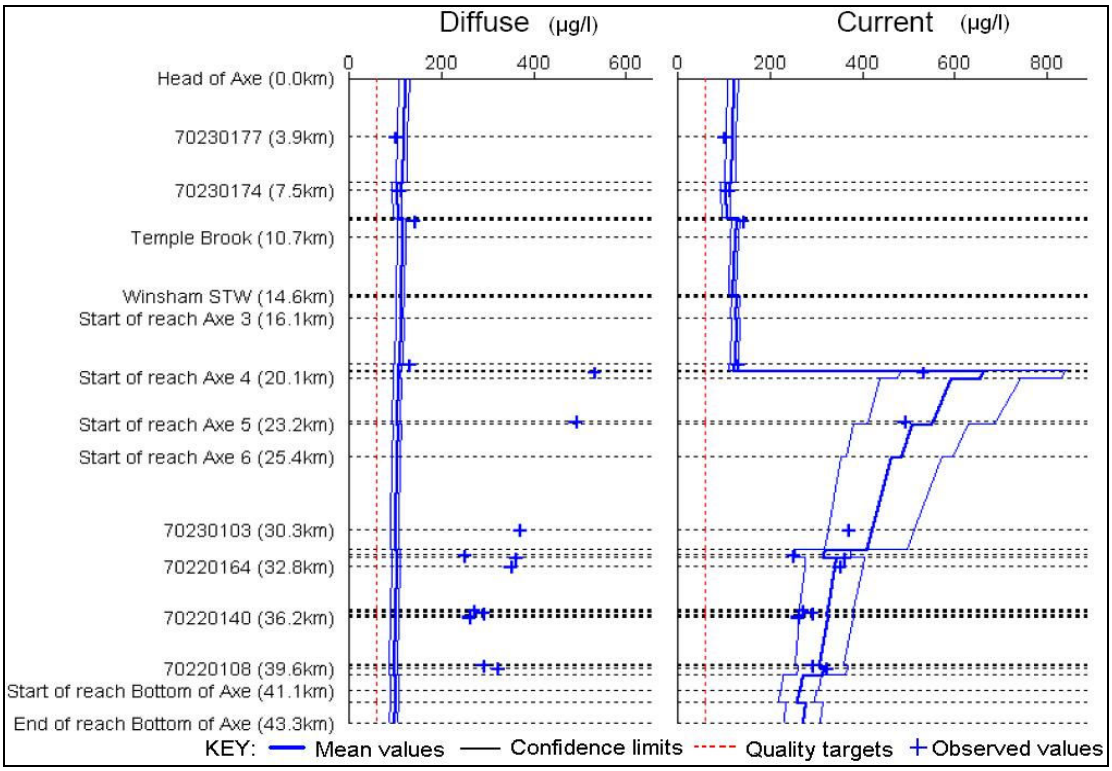


Figure 3.26: Comparison of 'current' situation and 'diffuse sources only' SIMCAT predictions for phosphorus levels in the River Axe. Observed values (+) have been retained in both scenarios for ease of comparison. The red dotted line denotes guideline standard. (Source: Ash et al., 2005)

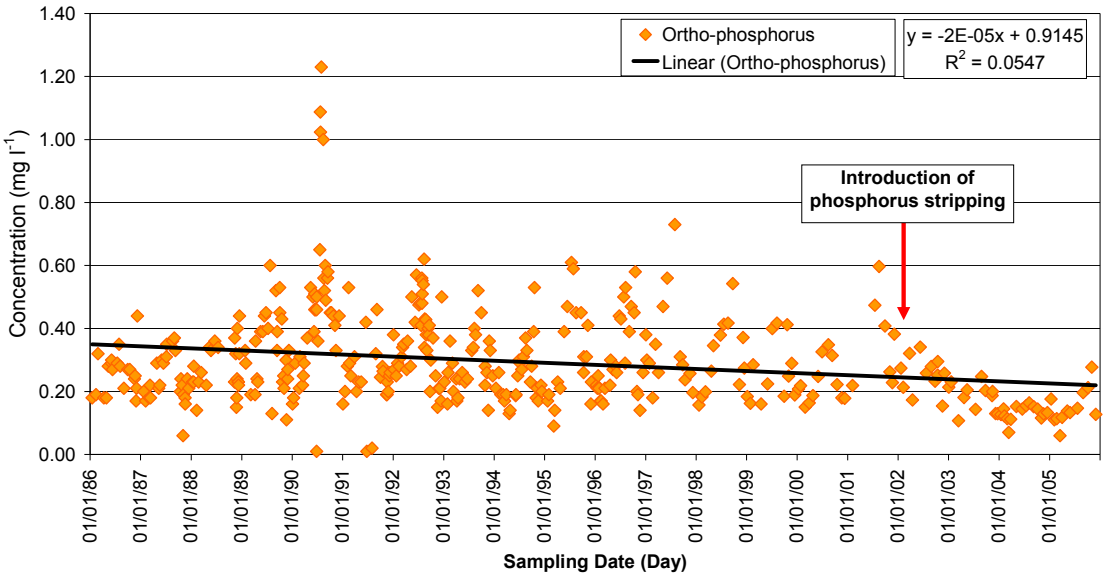


Figure 3.27: Measured concentration of ortho-phosphorus (PO_4^{3-}) at Whitford gauging station for the period from 1986 to 2006.

Sediment as the diffuse polluter in the river Axe catchment is the result of two processes, bank erosion and in surface runoff. Research by Eyquem (2006) revealed contrasting geomorphological characteristics of river banks between the upper Axe catchment, the upper Yarty tributary and lower courses. It was found that the upper Axe catchment watercourses are relatively stable, due to tree-lining and cohesive bank material. The upper reaches of the River Yarty are also tree-lined but fluvial erosion is present in toe scour and cliff erosion. The lower part of the catchment is adjusting laterally through meander growth and cut-offs. Bank erosion is being accelerated by poaching of banks by livestock, a lack of riparian vegetation and unprofessional restoration interventions by landowners. Palmer (2004), in his report on soil structural conditions in the Axe catchment, warns that machinery or high stocking levels are the main reason for compaction at the base of the plough layer and the loss of natural soil structure, causing a significant presence of surface run-off and erosion across whole fields.

3.1.8 Designations

Landscape designations include the Blackdown Hills Area of Outstanding Natural Beauty (AONB), East Devon AONB and West Dorset AONB, these cover the north, south and east parts of the catchment respectively. The Blackdown Hills have been designated as a Natural Area by English Nature (NAEN); and has been the subject of an Environmentally Sensitive Area scheme (ESA) managed by DEFRA.

The River Axe itself is primarily designated as a Special Area of Conservation (SAC) and there are several Sites of Special Scientific Interest (SSSI) on account of its importance as a river with distinctive communities of floating vegetation. Only the lower reaches of the main river have been designated, where the mixed catchment geology of sandstones and limestones gives rise to calcareous waters where water crowfoot *(Ranunculus penicillatus* ssp. *Pseudofluitans)* dominates, giving way to *Ranunculus fluitans* further downstream. Short-leaved water-starwort *Callitriche truncata* is an unusual addition to the *Ranunculus* community and gives additional interest (JNCC, 2006). This area is considered to support a significant presence of three

fish species of European importance; the Bullhead (*Cottus gobio*), Brook Lamprey (*Lampetra planeri*) and Sea Lamprey (*Petromyzon marinus*). Because of the above facts, the River Axe is one of the UK and Devon County high priority Habitats Directive sites (JNCC, 2006).

The Axe is also a priority catchment for the English Catchment Sensitive Farming Delivery Initiative (ECSFDI) between years 2006 – 2008, which was introduced by DEFRA in support of Water Framework Directive (WFD).

3.2 DATA COLLECTION, ANALYSIS AND MODEL BUILD

3.2.1 Model Requirements

For the purpose of this research SWAT-2005 and ArcView GIS 3.2 interface AVSWAT-X were used. SWAT is a comprehensive model that requires a wide diversity of information in order to run. It was developed to predict the impact of land management practices on water, sediment and agricultural chemical yields in large complex catchments with varying soil, land use and management conditions over long periods of time (Neitsch et al., 2005). To build the SWAT model of the Axe the following data were collected. Some data have been estimated from various sources as there was no direct source of required information for the study area. Sources of obtained data are shown in Appendix C1.

Spatial data

- Topography,
- Stream network,
- Outlet and inlet points,
- Main catchment outlet point,
- Land use map,
- Soil map and properties (types and physical data)

Temporal data

- Weather data (daily: rainfall, temperatures, solar radiation, wind speed, relative humidity and potential evapotranspiration)
- Land management practices (crop type, crop rotation, tillage, fertiliser, pesticide)
- Agricultural statistics (crop area, stock density)
- Water abstraction (water use) and discharges (point sources)

Calibration and Validation data

- River flow (discharge)
- Water quality (nitrate-nitrogen, ortho-phosphorus)

SWAT can simulate a single catchment or a system of hydrologically connected catchments. The first step in setting up the model is definition of main catchment outlet point and catchment subdivision and delineation into sub-units which can be based on different options: reach/main channel segments – one per sub-catchment; point sources; pond/wetland/reservoir – one per sub-catchment; land use or soil type – HRU; water quality and flow measurement points. At this point the stream network for the catchment and outlet and inlet points for sub-catchments are defined. Each sub-catchment contains at least one HRU, tributary channel and main channel. Sub-catchments are spatially related to each other.

The second step is land use and soil characterization and determining land use/soil class combinations for the delineated sub-catchments. In this process land use and soil classes are overlaid and combined into Hydrological Response Units (HRU-s). They are portions of a sub-catchment with unique land use/soil attributes and later with unique management. The use of HRU-s allows the model to reflect differences in evapotranspiration and other hydrological conditions for different land covers/crop - soil combinations. It also increases the accuracy of load predictions and provides better physical description of the water balance (Di Luzio et al., 2002). A very important step in defining HRU-s is the distribution command and assigning the number of HRU-s to each sub-catchment. This can be done by setting a minimum threshold area for land use

and soil classes to be included. A threshold area of 10%:10% has been used for the study area, to define HRU-s. Threshold area means that land use or soil class that covers less than the threshold level is eliminated in order to minimise the number of HRU-s whilst not overly compromising model accuracy. Different threshold areas give different HRU resolution within model. Table 3.2 gives an example for the study area using SWAT cover classes. With changing threshold area, the number of HRU-s is rising or falling.

Table 3.2: Relationship between threshold area, number of HRU-s and percentages of different land use types based on 25 sub-catchments of the river Axe catchment study area.

Threshold area Land use : Soil	Number of HRU-s	Land use type – SWAT (%)							
		AGRC	PAUK	FRSD	FRSE	FRST	WATR	URMD	MAGR
0:0	392	20.77	72.88	3.26	0.71	0.08	0.07	1.33	0.89
5:0	316	21.30	74.76	1.88	0.14	-	-	1.37	0.57
5:5	207	21.30	74.76	1.88	0.14	-	-	1.37	0.57
10:5	186	20.45	76.89	1.02	0.14	-	-	0.94	0.57
10:10	147	20.45	76.89	1.02	0.14	-	-	0.94	0.57
15:10	120	16.47	81.90	0.67	0.14	-	-	0.15	0.66
20:10	113	14.60	84.06	0.67	-	-	-	-	0.66
20:15	89	14.60	84.06	0.67	-	-	-	-	0.66

Key: AGRC - Agriculture Land -Crop; PAUK-Pasture; FRSD - Forest-Deciduous; FRSE - Forest-Evergreen; FRST - Forest-Mixed; WATR -water, URMD - Residential-Medium Density; MAGR - User Specified Grassland (Wildlife Park)

The third step is importing weather data and gauging station locations. Weather data is vital to the model as it drives the water cycle. For each type of required weather data, a location table is needed which is linked to the actual weather data. SWAT requires rainfall, temperature, solar radiation, wind speed and relative humidity. SWAT links to the nearest gauging station for each type of weather data for each sub-catchment.

The fourth step is creation of database files containing the information needed to generate default input for SWAT. Database files called input data are divided in to twelve files: Watershed Configuration File (.fig), Soil Input (.sol), Weather Generator Input (.wgn), Subbasin General Input (.sub), HRU General Input (.hru), Main Channel Input (.rte), Groundwater Input (.gw), Water Use Input (.wus), Management Input (.mgt), Soil Chemical Input (.chm), Pond Input (.pnd), Stream Water Quality Input (.swq). All collected and processed spatial and temporal data that defines the catchment characteristics are entered into these files.

The fifth step is input file modification through the Databases (Land Cover/Plant Growth, Users Soils, Users Weather Stations, Tillage, Fertilizer, Pesticide and Urban Area), Point Source Discharges (for each sub-catchment), Inlet Discharges, Reservoirs and Sub-basins data (editing input data for each HRU). It is often that default parameters (defined by SWAT) which can be found in above databases do not meet study area characteristics. Therefore, modifications are necessary and required to build a model which reflects the closest approximation of the real catchment conditions.

3.2.2 Sub-catchment Delineation

To define the river Axe catchment configuration a detailed 50×50 m grid digital elevation model (DEM) has been used (Figure 3.28). Due to the low elevation in the estuary part of the catchment, where SWAT was unable to adequately model ground levels at sea level, 1 m has been added to the DEM. As a result of this change all previously disconnected tributary channels became connected with the main river channel and had one whole catchment outlet point. This gave the opportunity to model diffuse nutrients at only one main catchment outlet and as one output file.

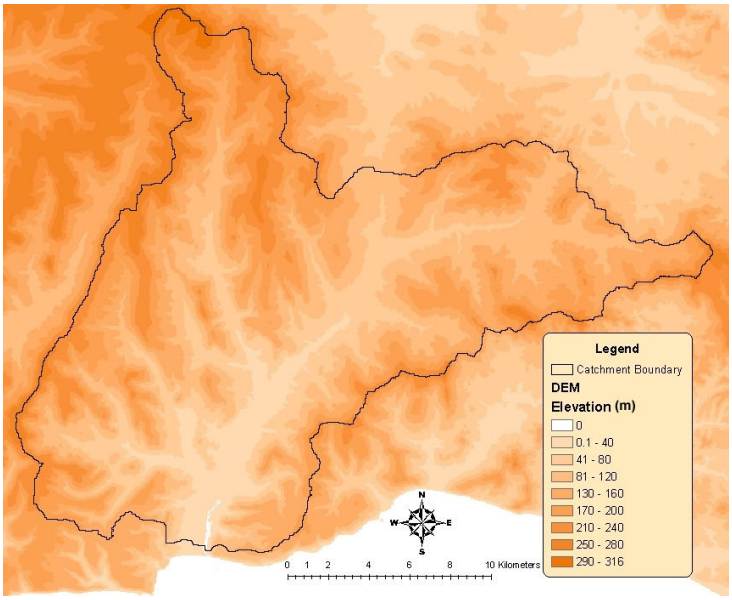


Figure 3.28: Digital Elevation model (DEM) – in metres above sea level.

The river network for the study area was defined by the model's automatic definition option (Figure 3.29). The threshold area, critical source area that defines minimum drainage area required to form the origin of a stream, has been set at 400 ha. This value returned the best representation of the rivers network. During the process of sub-catchment definition the majority of the automatically defined outlet points have been removed. Additionally the Environment Agency (EA) flow gauging station and water quality sampling points were used to define outlet points of sub-catchments.

The river Axe catchment has been delimited into 25 sub-catchments (Figure 3.29) which possess a certain geographic position in the catchment and are spatially related to each other. As can be seen from Figure 3.29, outflow from sub-catchment 1 enters sub-catchment 2. The sub-catchment delineation has been defined by the surface topography provided by the DEM. Each of the sub-catchments contains at least one hydrological response unit (HRU), a tributary and one main channel or reach (Neitsch, 2005). Within the river Axe catchment model each sub-catchment has between 2 and 10 HRU-s. For the representation of the results in later chapters, rather than each sub-catchment the groups of the sub-catchments were used based on the river or tributary channel that flows through them. Thus four sections were established (Figure 3.29) – the river Yarty section includes sub-catchments 21-24; the river Coly section includes sub-catchments 5-6, 12-14; the Upper Axe section includes sub-catchments 1-4, 15-20; the Lower Axe section includes sub-catchments 7-11 and 25.

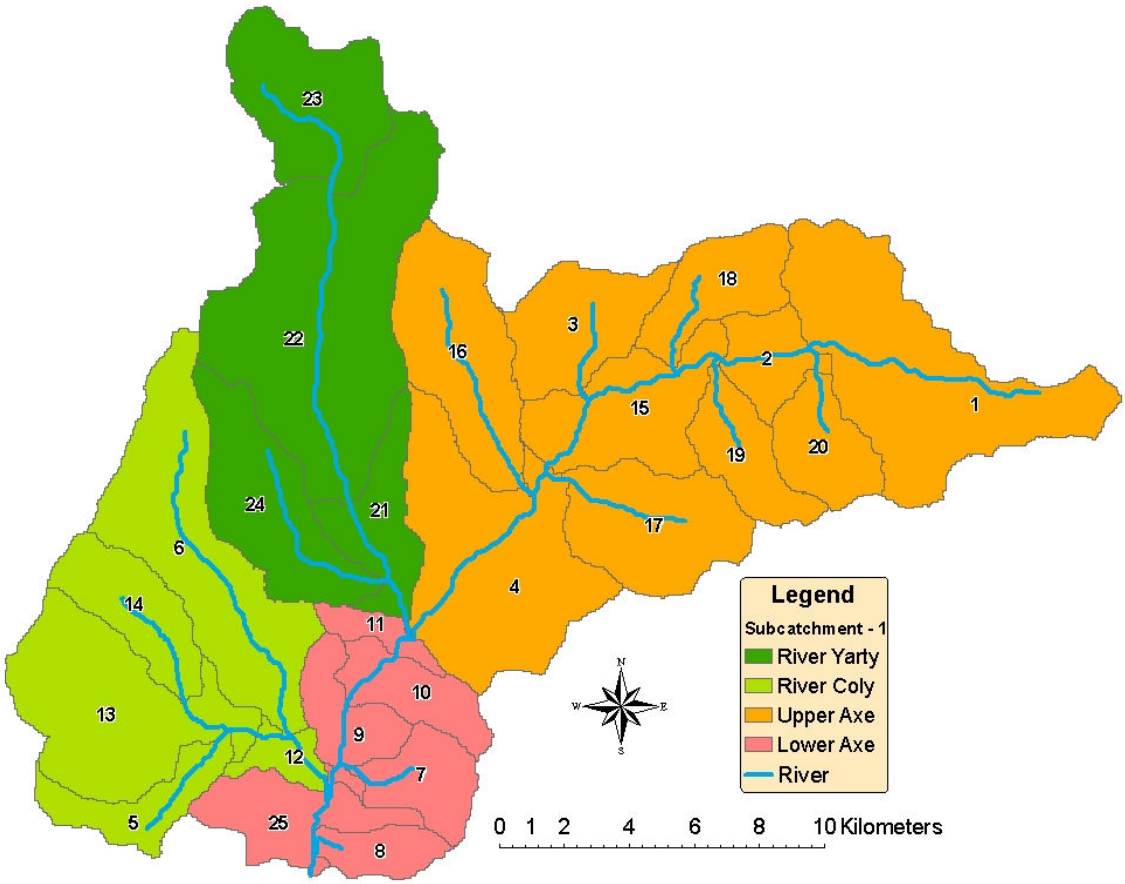


Figure 3.29: The river Axe catchment delineation into 25 sub-catchments and 4 catchment sections (River Yarty, River Coly, Upper Axe, Lower Axe).

3.2.3 Climate Data

The main climate variables required by SWAT are daily precipitation, maximum and minimum daily air temperature, relative humidity, solar radiation and wind speed. Most of this is observed data from meteorological gauging stations. A second option in SWAT is to us the weather generator to fill in missing data. All required data were obtained from the British Atmospheric Data Centre (BADC, 2006) (Figure 3.30). Daily precipitation (mm day^{-1}) has been used from nine gauging stations in the river Axe Catchment (Farway, Ford Abbey, Cricket St. Thomas, Chard Junction, Wambrook, Buckland, Seaton Junction, Wilmington, Pinhay). For the other climate parameters no gauging stations were in the catchment so the closest ones were used. Minimum and maximum daily air temperature (°C) has been obtained for four gauging stations (Lower Kingcombe, Sidmouth, Dunkeswell, Hemyock). Relative humidity (%) and wind speed

(knots) from two (Sidmouth, Hemyock) and solar radiation (sunshine hours) from one gauging station (Hemyock). As some climate data during the 20 year model period were missing they were replaced with data from the closest neighbouring gauging station.

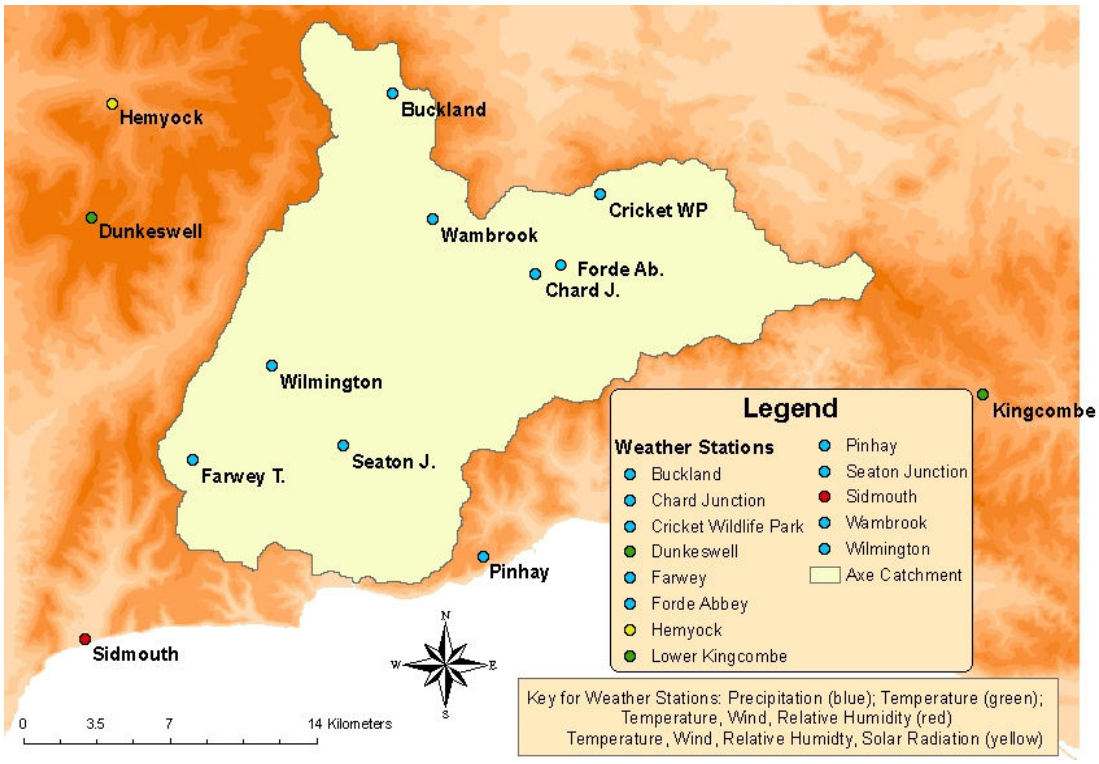


Figure 3.30: Weather gauging stations used in SWAT modelling of the study area.

WaSim-ET (v1.3) (Hess, 2000) has been used to calculate solar radiation (MJ m^{-2} day^{-1}) from hours of sunshine (hours).

Solar Radiation

Solar radiation was calculated using the Ångström equation, which relates incoming solar radiation to extraterrestrial radiation and relative sunshine duration (eq1):

Solar radiation (R_s):

$$R_s = R_a \left(a_s + b_s \frac{n}{N} \right) \qquad \textbf{(eq1)}$$

Where:
- R_s incoming shortwave radiation (MJ m^{-2} d^{-1})
- R_a extra-terrestrial radiation (MJ m^{-2} d^{-1}) (16.83 MJ m^{-2} d^{-1})
- N maximum daylight hours (hour) (11.98 hours)
- n actual duration of sunshine (hour)
- $\frac{n}{N}$ relative sunshine duration (hour)
- a_s Ångström regression constant (0.25)
- b_s Ångström regression constant (0.50)

Potential Evapotranspiration

Daily potential evapotranspiration (PET) values can be calculated by SWAT or read in by the user. The BADC data has been used to calculate PET. For the calculation of PET by SWAT the Penman-Monteith equation (eq2) was selected. It combines components that account for energy needed to sustain evaporation, the strength of the mechanism required to remove the water vapour and aerodynamics and surface resistance terms (Neitcsh et al., 2005). PET using Penman-Moteith equation is calculated using four different climate data: temperature (min, max), solar radiation, relative humidity and wind.

Potential evapotranspiration (MJ m^{-2} d^{-1}):

$$\lambda \mathrm{E} = \frac{\Delta \cdot \left(H_{net} - G\right) + \rho_{air} \cdot c_p \cdot \left[e_z^{\circ} - e_z\right] / r_a}{\Delta + \gamma \cdot \left(1 + r_c / r_a\right)} \qquad \textbf{(eq2)}$$

Where:
- λE latent heat flux density (MJ m^{-2} d^{-1})
- E depth rate evaporation (mm d^{-1})
- Δ slope of the saturation vapour pressure-temperature curve de/dT (kPa $°C^{-1}$)
- H_{net} net radiation (MJ m^{-2} d^{-1})

G heat flux density to ground (MJ m^{-2} d^{-1})

ρ_{air} air density (kg m^{-3})

c_p specific heat at constant pressure (MJ kg^{-1} $°C^{-1}$)

e_z° saturation vapour pressure of air at height z (kPa)

e_z water vapour pressure of air at height z (kPa)

γ psychometric constant (kPa $°C^{-1}$)

r_c plant canopy resistance (s m^{-1})

r_a diffusion resistance of the air layer (aerodynamic resistance) (s m^{-1})

3.2.4 Land use

The CORINE 2000 land cover map has been used to provide a spatial distribution of major land cover classes within the study area (CLC, 2000). This land cover map defines arable land within one class, pastures within another class and so on. To get a better representation of arable land, which was calculated from EDL (2006) data, three CORINE land cover map classes within agricultural areas (211, 242 and 243) were combined in just one class as arable. Further on SWAT land use classes were assigned to original cover classes of CORINE map (Table 3.3). Figure 3.31 presents the modified land use map used in SWAT.

Table 3.3: Relationships between CORINE and SWAT land use classes used in model.

SWAT land use classes	CORINE land cover classes in the river Axe catchment
AGRC - Arable	211 – Non-irrigated arable land 242 – Complex cultivation patterns 243 - Land principally occupied by agriculture, with significant areas of natural vegetation
PAUK - Pasture	231 - Pastures
MAGR – Wildlife Park	142 – Sport and leisure facilities (Wildlife parks)
FRSD – Deciduous forest	311 – Broad-leaved forest
FRSE – Evergreen forest	312 – Coniferous forest
FRST – Mixed forest	324 – Transitional woodland - shrubs
WATR - Water	423 – Intertidal flats
URMD - Urban	112 – Discontinuous urban fabric

To define hectares of certain crops and rotations within the study area arable land, EDL Agricultural Census (EDL, 2006) with 2km grid resolution data for 1969-2004, have been used.

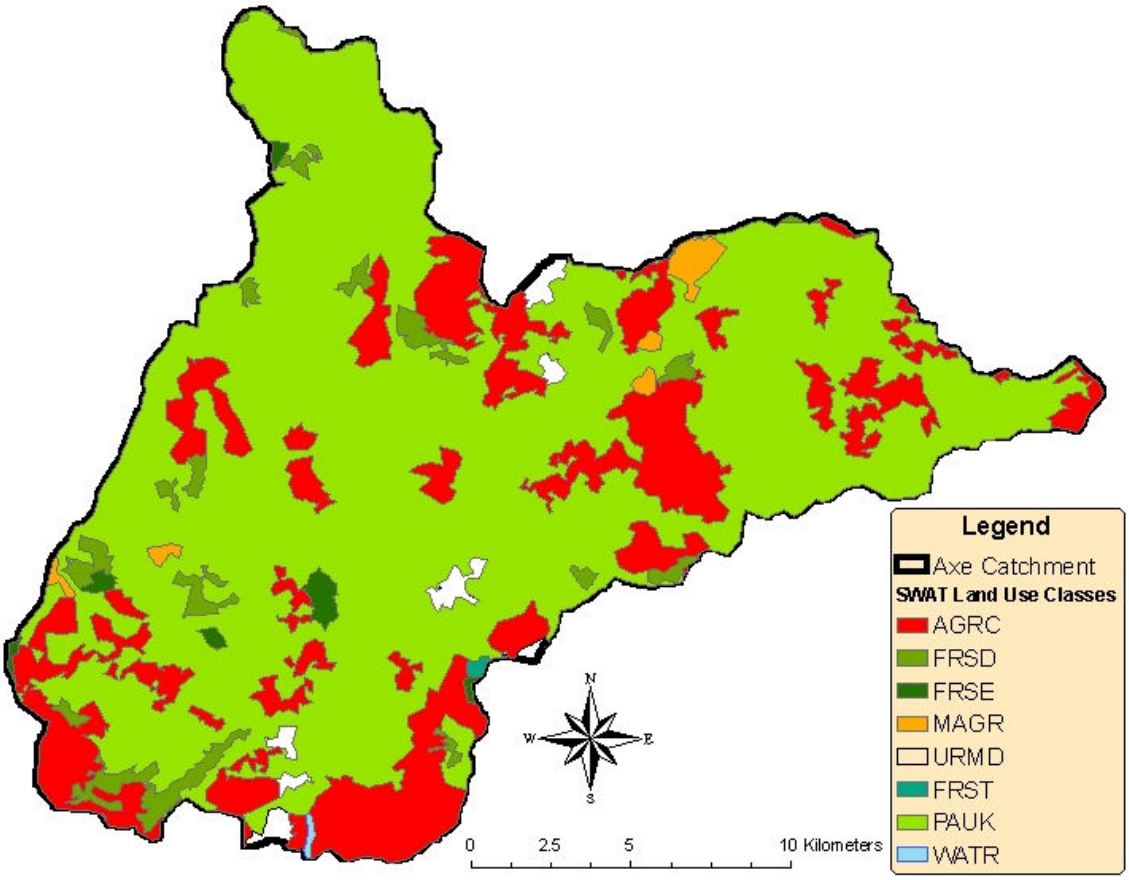


Figure 3.31: Modified land use map used in SWAT modelling of the study area.

3.2.5 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 (Neitsch et al, 2005). The soil input (.sol) files defines the physical properties for all layers in the soil.

The digital National Soil Map (NATMAP, 1999) at scale 1:250.000 has been used to define soil types (Figure 3.5, Appendix A1). The NATMAP details the distribution of 300 soil association 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 National Soil Resource Institute (NSRI) SEISMIC (1994) database provides soil characteristic data and has been utilised to gain the appropriate inputs for SWAT. SEISMIC data are based on an average of 2 to 3 soil observations nationally. Soil data provided by SEISMIC does not represent soils on a local scale but is based on a small number of national samples and predicted values. As Whitehead (2006) suggest this may affect the accuracy of the soil modelling within SWAT.

SWAT requires general values for the following parameters: soil name; number of layers; hydrological group; rooting depth; anion exclusion factor (default) and potential crack volume (optional). A further set of parameter information is required for each layer: depth from surface; depth of obstacle to roots; bulk density; available water content; saturated hydraulic conductivity; percent of organic matter, clay, silt, sand, rock; albedo and erodibility factor (K_{USLE}). Appendices D1 to D6 contain soil parameter values used in the model.

Two parameters (soil albedo, K_{USLE}) required by SWAT are not included in the NSRI SEISMIC database and have to be obtained from other sources or calculated using other soil parameters.

Soil Albedo

Soil albedo is function of soil colour (Findlay et al, 1984) and angle of incidence of solar radiation and depends on the inherent colour of the parent material, organic matter content and weathering conditions (NRCS, 2005) (eg3).

Soil Albedo: $\text{Albedo} = (0.07 \times \text{colour value}) - 0.12$ **(eq3)**

Soil erodibility factor

Soil erodibility factor was calculated using the alternative Universal Soil Loss Equation (K_{USLE}) (eq4, eq5, eq6, eq7, eq8) proposed by Williams (1995).

Soil erodibility factor: $$K_{USLE} = f_{csand} \cdot f_{cl-si} \cdot f_{orgc} \cdot f_{hisand} \quad \textbf{(eq4)}$$

Where: f_{csand} factor that gives low soil erodibility factors for soils with high coarse sand contents and high values for soils with little sand,

$f_{cl\text{-}si}$ factor that gives low soil erodibility factors for soils with high clay to silt ratios,

f_{orgc} factor that reduces soil erodibility for soils with high organic carbon content,

f_{hisand} factor that reduces soil erodibility for soils with extremely high sand contents.

The factors are calculated as:

$$f_{csand} = \left(0.2 + 0.3 \cdot \exp\left[-0.256 \cdot m_s \cdot \left(1 - \frac{m_{silt}}{100}\right)\right]\right) \quad \textbf{(eq5)}$$

$$f_{cl-si} = \left(\frac{m_{silt}}{m_c + m_{silt}}\right)^{0.3} \quad \textbf{(eq6)}$$

$$f_{orgc} = \left(1 - \frac{0.25 \cdot orgC}{orgC + \exp[3.72 - 2.95 \cdot orgC]}\right) \quad \textbf{(eq7)}$$

$$f_{hisand} = \left(1 - \frac{0.7 \cdot \left(1 - \frac{m_s}{100}\right)}{\left(1 - \frac{m_s}{100}\right) + \exp\left[-5.51 + 22.9 \cdot \left(1 - \frac{m_s}{100}\right)\right]}\right) \quad \textbf{(eq8)}$$

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

3.2.6 Management Files

The primary goal of management files is to assess the impact of human activities on a given system. Central to this assessment is the itemisation of land and water management practices taking place within the system. The primary file used to summarize these practices is the HRU management file (.mgt). This file contains input data for planting, harvest, irrigation, nutrient applications, pesticide applications and tillage operations (Neitsch et al., 2005). The user can specify the beginning and the end of the growing season, schedule the management operations (harvest, grazing) occurring at specific times, type and amount of fertiliser, pesticide and irrigation applications and type of tillage operations.

Crop Rotations

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 number of crops grown within one year in the HRU, but only one crop can be grown at once.

Arable land represents 20% of the study area of which more than 50% is seeded with temporary grassland. To set a baseline scenario, originally typical rotations for the south-west region of the UK were used to vary crops grown from year to year within the SWAT model (Table 3.4). As the goal has not been to simulate actual management (it is impossible), but to define representative management operations, five different rotations used in the model have been applied across the study area arable land. Two (ROT1 and ROT3) five year rotations have been taken from National Groundwater Recharge

Assessment – Crop Calendar Dataset (Holman et al., 2005), which are based on ADAS standard rotation information for soil characteristics in South West England. The other three (ROT2, ROT4, ROT5) have been obtained during interviews with the Environment Agency and farmers on a field trip to the river Axe catchment. Exact information on plant and harvest dates was not available for the Axe catchment study area. Therefore, to create management files for SWAT, indicative crop calendar dates were used (Holman et al., 2005). Appendices E1 to E5 present crop management dates as input to the SWAT base scenario model.

Table 3.4: Percent (%) of arable and total study area covered by crop rotations in the model.

Crop Rotation	Crops	Arable area (%)	Study area (%)
ROT1	rg/rg/ww/ww/wb	31.86	6.51
ROT2	m	12.77	2.61
ROT3	rg/rg/ww/ww/sa	26.89	5.50
ROT4	m/wb/rg	9.44	1.93
ROT5	rg/rg/rg/rg/rg	19.05	3.89

Key: rg = rye grass, ww = winter wheat, wb = winter barley, m = maize, sa = set-aside

The percentage of a particular rotation in the study area was defined and based on EDL (2006) data for the year 2000. Percentages of particular crops obtained from the database for the year 2000 gave a good representation to set rotations. To calculate the total area of arable land, the area of five (EDL, 2006) crop classes (winter wheat, winter barley, maize, set-aside, temporary grass) which are represented in the selected rotations, were extracted for the study area. The sum of these crop areas (8,875 ha) is close to the area of arable land in the CORINE land cover map (8,305 ha) and in SWAT after HRU-s distribution (8,191 ha) (Table 3.5, Figure 3.32). There is a noticeable discrepancy of 684 ha between actual and SWAT arable area, mainly due to the principle of collecting and presenting agricultural statistics by EDL (2 km grid square) and the low resolution land use map (1:250,000).

Table 3.5: Percent (%) and area (ha) of crops used in SWAT arable rotations.

Crop	Rotations										Total crop (ha)	Total crop (%)
	ROT 1		ROT 2		ROT 3		ROT 4		ROT 5			
	ha	%	ha	%	ha	%	ha	%	ha	%		
Winter wheat	843	33			733	33					1577	19
Winter barley	422	17					287	33			708	9
Maize			1021	100			287	33			1308	16
Set-aside					367	17					367	4
Rye grass	1265	50			1100	50	287	33	1580	100	4232	52
Total	2530	100	1021	100	2200	100	860	100	1580	100	8191	100
Percent of arable area (%)	**31**		**12**		**27**		**10**		**19**		**100**	/

Rotations were further assigned to specific soil types based on ADAS soil texture classes, RB209 - Fertiliser recommendations (MAFF, 2000) and Soil Survey of England and Wales for South West England (Findlay et al., 1984) recommendations for soil use and land cultivation. Combinations between arable HRU-s and rotations are presented in Appendix F1. The exact agreement between calculated areas of each rotation and actual was difficult to achieve as certain soil series are not suitable for all rotations.

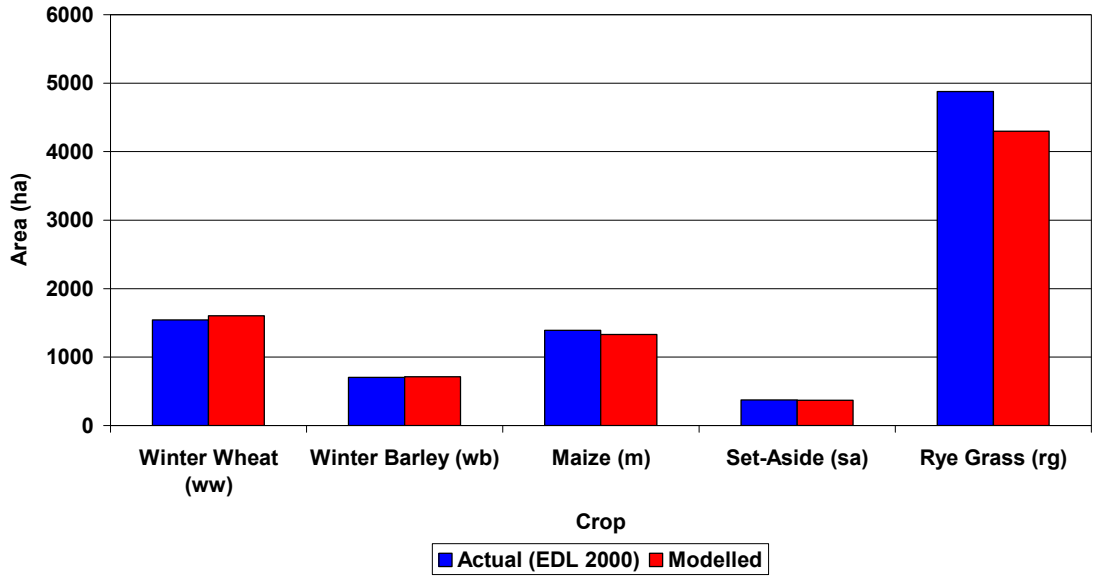


Figure 3.32: Comparison of actual and modelled areas of crops in the study area.

An important SWAT parameter is the harvest index override ((kg ha^{-1}) (kg $ha^{-1})^{-1}$) (HI_OVR) in harvest operations (*.mgt). This variable forces the ratio of yield to total above ground biomass to the specified value. The harvest index in the plant growth database (crop.dat) assumes only the seed is being harvested. If biomass is cut and removed (hay cuttings, grass silage, maize silage), HI_OVR must be used to specify the amount of biomass removed. A value of 0.95 was used for the baseline and future scenarios.

Pastures and Grazing

SWAT pasture land use class represents more than 76% of the study area and taking in to consideration that dairying is the major agricultural practice it can be clearly seen that setting management files for pastures is an important step in building the model. To set a baseline scenario five different rotations (grazing and grass cut) have been used, based on interviews with farmers and site observations. Appendices E6 to E8 represents management operation dates as input to the SWAT base scenario model. Pasture area was first divided based on livestock units (LU) density of sheep and dairy cows. As literature suggests an average density of 4 LU dairy cow and 2.25 LU sheep per hectare of pasture (Frame, 1992; Haines, 1982), the total area of dairy cow and sheep pastures was calculated to fulfil this requirements.

Sheep are rotated between two grazing pastures (11%) of which each represent approximately 7.5% of the pasture area. Sheep were grazing for 180 days from 1st of April to 30th of September, with an average stocking density of 2.25 LU per ha of pasture or 4.5 LU per ha of paddock and grazing period of 30 days per paddock. Dairy cows were also rotated, between one pasture and one cut grass silage field harvested in May and later grazed, of which each represents approximately 8.5 % of the pasture area. The grazing period lasted 183 days, from the 1st April to 3rd October, with grazing period per paddock 60 days (April to May) and 30 days (June to October). Average stocking density was 4 LU per ha of pasture or 8 LU per ha of paddock. The fifth class is three cuts grass silage field with no grazing, and represents approximately 67% of the study area pastures (Table 3.6).

Table 3.6: Percent (%) of pasture and total study area covered by pasture rotations in the model.

Rotation	Management	Pasture Area (%)	Study Area (SWAT) (%)
PA11	sheep grazing	7.59	5.84
PA12	sheep grazing	7.61	5.85
PAS2	dairy cow grazing	8.45	6.50
PAS3	dairy cow grazing	8.65	6.65
PSG4	3-cut grass silage	67.70	52.06

Grazing and grass systems were further assigned to specific soil types based on ADAS soil texture classes, RB209 - Fertiliser recommendations (MAFF, 2000) and Soil Survey of England and Wales for South West England (Findlay et al., 1984) recommendations for soil use and land cultivation. Combinations between pasture HRU-s and grass grazing/cut systems are presented in Appendix F2. The exact agreement between calculated and actual areas of each system was difficult to achieve as certain soil series are not suitable for all systems.

Crucial variables required by SWAT to be entered in management files for grazing operations are the number of grazing days (GRZ_DAYS), dry weight of biomass consumed daily (BIO_EAT), dry weight of manure deposited daily (MANURE_KG) and minimum plant biomass below which grazing is not allowed (BIO_MIN). Number of grazing days was obtained from various literature sources: MAFF (2000) suggest 28 days, FAPRI (2006) which also used the SWAT model suggest one month (30 days), NE (1999) suggests 3-4 weeks, Frame (1992) suggests 10 (20) to 30 days depending on stocking rate and on the time in the year. For a better and easier overview of the models management files 30 day grazing periods were used on one paddock.

Dry weight biomass consumed daily and manure deposited daily has been calculated using average number of livestock units in the study area and average values for biomass consumption and manure deposition for dairy cows and sheep (Table 3.7). Information on number of livestock units was extracted from EDL (2006) data for the year 2000 and multiplied with LU factors adapted to study area (1 cow (600 kg) = 1 LU; 1 sheep (75 kg) + lamb (30 kg) = 0.15 LU). BIO_MIN can vary according to the condition of the pasture. FAPRI (2006) suggests that pastures in good condition are

assumed to have at least 7.62 cm of grass to allow grazing (700 kg ha^{-1} on dry basis) and pasture in poor condition were assumed to have 2.54 cm (200 kg ha^{-1} on dry basis). For the baseline scenario a value for poor condition pastures has been selected as field research in the study area showed poor condition of pastures with high surface runoff. Information on biomass eaten and manure deposited was gathered from different sources (Frame, 1992; DEFRA, 2007-b; NE, 2007; NDGMT, 1998; Chanasky et al., 2002) and some recalculations were done to meet SWAT requirements and to fulfil study area characteristics (Table 3.7 and 3.8).

Table 3.7: Adopted average grazing parameters included in SWAT management files (Adapted from: Frame, 1992).

	LU	Animal weight (kg) adopted	Biomass requirements (DM kg day^{-1} LU^{-1})	Manure deposited (DM kg day^{-1} LU^{-1})
Dairy cow	1	600	16.50	3.75
Sheep	0,15	75	2.47	0.82

Key: LU – livestock unit; DM – dry matter

Table 3.8: Grazing parameters for dairy cow and sheep per HRU included in SWAT grazing management files.

Animal type	BIO_EAT	MANURE_KG	BIO_MIN
	kg DM ha^{-1} day^{-1}	kg DM ha^{-1} day^{-1}	kg DM ha^{-1} day^{-1}
Dairy cow (8LU)	132.00	30.00	200
Sheep (4.5LU)	74.10	24.60	200

Key: LU – livestock unit; DM – dry matter

Fertiliser Application

This 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, the amount of fertiliser/manure applied and the depth distribution of fertiliser application. SWAT assumes surface runoff interacts with the top 10 mm of soil. Nutrients contained in this surface layer are available for transport to the main channel in surface runoff (Neitsch et al., 2005). This indicates the high importance of correct fertiliser and manure application operations.

Information on fertilisation of crops and pastures has been gathered from a number of different sources. Application dates, fertiliser type and application rates have been taken from best practice guidelines (MAFF, 2000; ABC, 2006; SAC, 1999), published reviews (Hough, 1990; Bunting et al., 1978; Whitehead, 1995) and a PhD thesis (Whitehead, 2006). Both fertiliser types were used, mineral for arable land rotations (UREA – 46 % N, CAN – 27 % N and TSP – 20.7 % P) and for pastures organic and mineral fertiliser (dairy and sheep fresh manure, UREA – 46%) (Appendix G1). For the SWAT model crop growth it is not important which mineral fertiliser (elemental or UREA) is used just that the right amount is added to the crop. On the other hand it is important if manure is used because additional organic matter is added which slows the release of nutrients from organic matter. In this study the SWAT default values for dairy and sheep fresh manure were used. One of the SWAT shortcoming is that slurry application cannot be considered in the model at present. Fertiliser application rates can be reviewed in Appendices E1 to E5 for crop rotations and E6 to E8 for grazing/grass cut rotations.

3.2.7 Abstraction Data

Water abstraction is an important factor for the model because it can greatly affect the water balance, especially if water is removed from the catchment. SWAT uses a special management tool (*.wus) for consumptive water use that removes water from the catchment. This file is used to simulate removal of water for irrigation outside the catchment or removal of water for urban/industrial use. Water can be removed from the shallow aquifer, the deep aquifer, the reach or the pond on a sub-catchment level. 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 has to be specified (Neitsch et al., 2005).

Data obtained from the EA shows that within the river Axe catchment there are 51 licensed abstraction points. Figure 3.33 shows the location of abstraction points in the study area. These have been derived from the EA's database of active licensed abstraction permits for January 2006. All current licences/abstraction permits are

supplied with annual authorised abstraction quantity in cubic metres, and are the maximum for the point as a whole. Annual authorised return quantity for a licence as a whole is also supplied. All abstraction points are given grid references as NGR-s. A single abstraction point has one grid reference, a reach on a watercourse or well point system has 2 grid references, and an area (water may be taken at any point within an area marked on map) will have 4 grid references.

As all abstracted water is not removed from the catchment or used for urban/industrial use, some of the abstraction licences were not used in the model. This permit refers to mills as water is only diverted and to ponds and water removal from tidal area as they are not important (small quantities). Finally 46 licensed abstraction permits (136 abstraction points) have been used in the model.

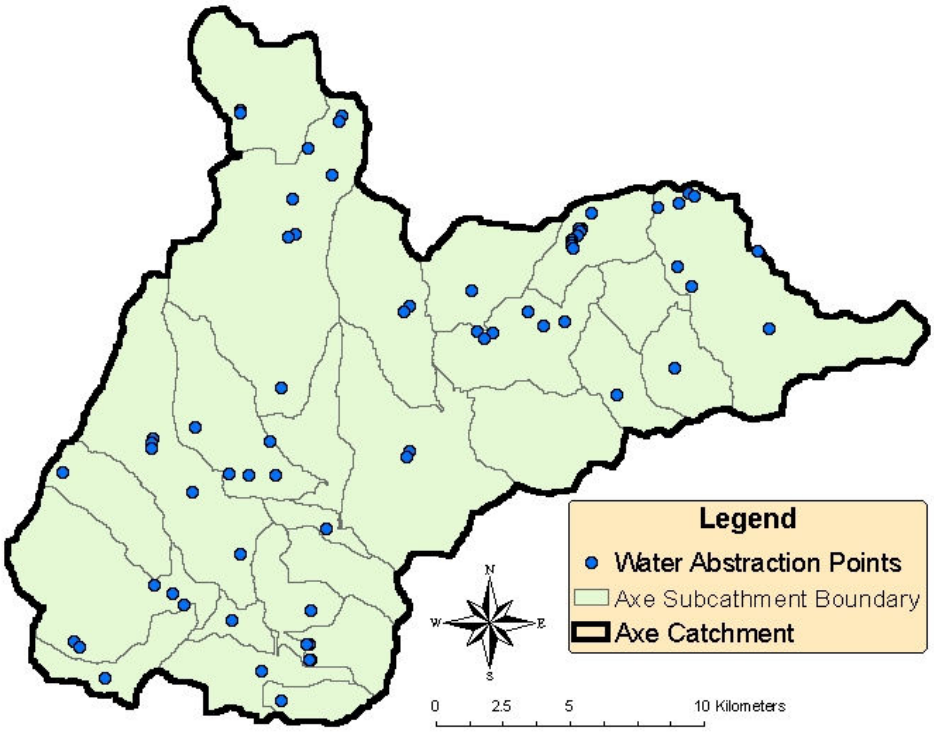


Figure 3.33: Locations of water abstraction points in the river Axe catchment.

As SWAT only allows an average daily volume of water to be removed from each sub-catchment, the licensed abstraction quantity for each abstraction point within an individual sub-catchment has been totalled to give one value per sub-catchment (Appendix H1). Data has been totalled separately for ponds, reaches, shallow and deep

aquifers as the model requires. Data has been entered in to the SWAT model through the Sub-basin Inputs dialog box.

An important fact is the quantity of actual water abstracted per authorised licence. Data from the EA suggest that at most, only 60% of the licensed quantity is used (1990-1996), even in dry years, but also predicts a future 20% rise in abstraction mostly because of new licenses (Morris et al., 1997). On the other hand Marechal (2004) in work carried out in East Anglia suggested that the actual amount of water abstracted is approximately 80% of licensed volumes. As a result of this authorised abstraction quantity has been reduced to 80% and applied to the SWAT model.

3.2.8 Discharge Data

To simulate the loadings of water and pollutants from sources not associated with a land area (e.g. sewage treatment plants), SWAT allows point source information to be read in at any point along the channel network. The point source loadings may be summarized on an hourly, daily, monthly, yearly, or average/constant annual basis. SWAT will read in water flow, sediment, nutrients, CBOD, dissolved oxygen, chlorophyll-a, pesticide, metals, and bacteria data from the point source files (Neitsch et al., 2005).

Information on active point sources discharges was obtained from the EA database. There are 401 point discharges in the river Axe catchment (Figure 3.34), 38 of them represent more than 98% of all daily flow discharged from point sources. The authorised quantity of effluent discharged is supplied as daily dry weather flow or effluent maximum daily flow both in cubic metres (m^3). As SWAT requires average daily flow all obtained values were recalculated. Where daily dry weather flow was given the values were multiplied by 1.24, and where maximum daily flow was given the values were estimated to be 50% to get average daily flow, as suggested by the EA. Actual flows were used where given.

When estimating average daily loading of other required parameter to model nutrient cycles such as sediment, organic nitrogen, organic phosphorus, nitrate, ammonia, nitrite,

mineral phosphorus, CBOD, dissolved oxygen and chlorophyll-a, actual measured data supplied from EA were used, or different literature was used to estimate appropriate values where not measured (Brake, 1998; Twichell et al., 2002; Salih, 2004; USGS, 2003; Clark et al., 2005; Neal et al, 2005; Ash et al., 2005). Some of the biggest sewage treatment works have been supplied with some of the parameters but not with all. There were no data for small sewage treatment works, for trade discharges and for fish farms. Daily loading for fish farms were estimated with use of published reviews and literature (Kendra, 1989; EPA, 2002; Papatryphon et al., 2005; Maillard et al., 2005; Helfrich and Libey, 2007). After data was set it was summarised to sub-catchment level. Values used in the SWAT model and calculated average daily loadings are presented in Appendices I1 to I3. Further on, data was entered in to the SWAT model in two ways. Firstly a table with discharge point locations was entered to the model during Watershed Delineation to distribute inlet points over sub-catchments. Secondly data was entered during input modification, through the Point Discharge dialog box as Constant Daily Loadings for each sub-catchment.

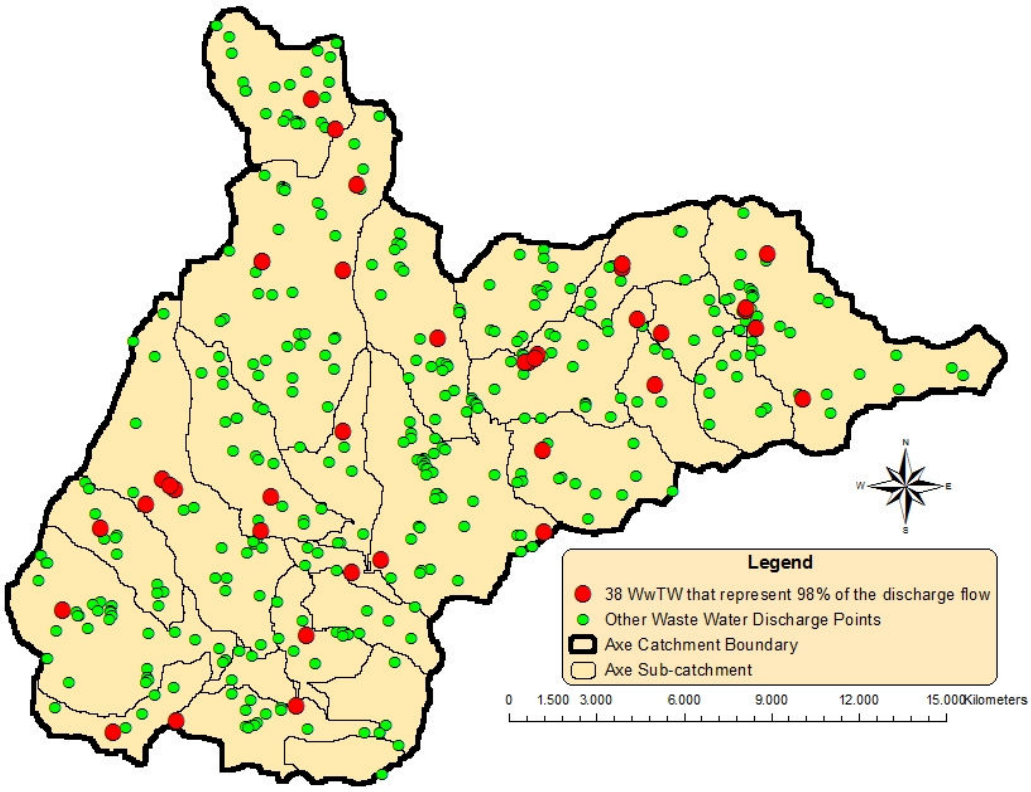


Figure 3.34: Locations of waste water discharge points in the river Axe catchment.

4 CALIBRATION AND VALIDATION

Many of the model's input parameters cannot be measured for different reasons, such as high cost of equipment or lack of time or personnel which means that the model must be calibrated. To ensure effective results for future land use management impacts on reduction of diffuse source nutrients, the model was firstly calibrated to measured data and then tested - validated (no further parameter adjustment) against an independent set of measured data.

Calibration means varying the coefficients of the designed model within an acceptable range until a satisfactory agreement between measured and computed output values is achieved. The variable to which the model is most sensitive should be calibrated first. The values of input variables, such as climate and land use/management, are needed for calibration. The data obtained from standard databases, that collected by different agencies, or the data measured in the field is used at this stage (Shirmohammadi et al., 2000). Validation is done by running the model with the coefficients established during calibration and with input corresponding to another data set or time period.

Validation is the assessment of accuracy and precision, and a thorough test of whether a previously calibrated parameter set is generally valid. In other words, validation in a strict sense requires that no input parameters should be obtained via calibration. It involves both operational and scientific examination. The scientific component should assess the consistency of the predicted results with the prevailing scientific theory. It may not be perfect in the case of empirical models. The evaluation should be done through statistical analyses of observed and predicted data. The model performance is accepted if there is no significant difference between the observed and predicted data (Shirmohammadi et al., 2000).

4.1 MODELLING STRATEGY

Important components of the water balance for correct representation of diffuse nutrient transport and losses are evapotranspiration, infiltration (lateral, groundwater flow) and surface run-off. As evapotranspiration is a function of crop growth only a proper simulation of crop growth and management can ensure realistic modelling of evapotranspiration and nutrients within a river catchment. Infiltration of nutrients through the soil profile depends on the quantity of water entering and moving through the soil profile in terms of percolation. Properly modelled infiltration will ensure accurate simulation of nutrient leaching to the water body (groundwater, river). If nutrient runoff is to be modelled, appropriate modelling of surface runoff is vital. Before nutrient modelling is attempted a correct partitioning of water in these three phases is required, apart from the requirement for a match of predicted and observed stream flow (Whitehead, 2006). To proceed with reliable calibration in SWAT it has to be done (if possible) in three stages (water balance and stream flow, sediment and nutrients). For the study area only river flow at Whitford gauging station and nutrient measurements (nitrate nitrogen, ortho-phosphorus) for several points on the river were obtained from the EA; Whitford gauging station was used for calibration purposes, whilst other points are also used in validation.

The model has been run for a period of 20 years, from the 1st January 1986 until the 31st December 2005. This long time period is important for smoothing the effects caused by extreme meteorological events (storms) or to exclude effects of dry or wet periods.

The study period was divided into three periods: warm up, calibration and validation. The warm up period extended over 2 years from 1st of January 1986 until 31st of December 1987, as this period is essential for stabilisation of parameters as the initial results can vary significantly from the observed values during. In this period the model initialises and then approaches reasonable starting values for model state variables and adjusts for inaccurate initial conditions (Whitehead, 2006). In this period the model deposits sediment in the river network and fills the soil profile with water before simulation results can be considered realistic. All outputs from this period of time are

excluded from statistical analysis. Data from 1st of January 1988 until 31st of December 1997 have been used in the calibration of the river Axe catchment model, as average annual precipitation (1018mm) in this period is closely matched to average annual precipitation over the whole period of 20 years (1063mm). Data from 1st of January 1998 until 31st of December 2005 have been used in validation of the model. The validation period is in fact wetter (1120mm) than the calibration period. Usually it is better to calibrate on wetter periods and validate on dryer periods. The twenty year study period showed changing hydrological behaviour (precipitation, river flow) in the catchment towards higher river flows for the same precipitation. Table 4.1 shows the data available and periods of data used for the calibration and validation exercise within the river Axe catchment model.

Table 4.1: Calibration and validation data available for the study area.

Parameter	SWAT equivalent output parameter (*.rch)	Calibration		Validation	
		Dates	Source	Dates	Source
Flow (m^3s^{-1})	FLOW_OUT	1988-1997	EA	1998-2005	EA
Nitrate nitrogen (NO_3-N)	NO3_OUT	1988-1997	EA	1998-2005	EA
Ortho-phosphorus (PO_4^{3-})	MINP_OUT	1988-1997	EA	1998-2005	EA

Calibration has been performed manually with the purpose of observing and investigating the impacts of SWAT parameters on simulation results. During the calibration process SWAT hydrological and nutrient parameters were changed to get better performance statistics. All the main parameters were chosen for calibration on the basis of previous SWAT studies (Kannan, 2003, Whitehead, 2006, Tavares, 2006) and the SWAT Calibration Techniques Manual (SWAT, 2007). These indicated where parameters have been found to be sensitive or were suggested as targets for change.

4.2 MODEL PERFORMANCE STATISTICAL CRITERIA

Simulated and observed time series of river flow, and nutrients were compared, for calibration and validation periods. A number of statistical tests can be considered in model performance evaluation. The four numerical model performance measures used are coefficient of determination (R^2 coefficient), Nash-Sutcliffe simulation efficiency

(E_{NS}) (Nash and Sutcliffe, 1970), root mean square error (RMSE) and percentage bias (PBIAS). All of these tests are generally known and widely used in hydrological model performance evaluation.

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

The coefficient of determination (R^2) for n time steps is calculated as (eq9):

$$R^2 = \left(\frac{\sum_{i=1}^{n} (simulated_i - simulated_{average})(measured_i - measured_{average})}{\sqrt{\sum_{i=1}^{n} (simulated_i - simulated_{average})^2} \sqrt{\sum_{i=1}^{n} (measured_i - mesured_{average})^2} } \right)^2 \quad \textbf{(eq9)}$$

It describes the portion of total variance in the measured data that can be explained by the model. The range is from 0.0 (poor model) to 1.0 (perfect model). A value of 0 for R^2 means that none of the variance in the measured data is replicated by the model, and value 1 means that all of the variance in the measured data is replicated by the model predictions. The fact that only the spread of data is quantified is a major drawback if R^2 is considered alone. A model which systematically over- or under predicts all the time will still result in good values close to 1.0 even if all predictions were wrong.

The Nash-Sutcliffe simulation efficiency index (E_{NS}) for n time steps is calculated as (eq10):

$$E_{NS} = 1 - \left(\frac{\sum_{i=1}^{n} (measured_i - simulated_i)^2}{\sum_{i=1}^{n} (simulated_i - simulated_{average})^2} \right) \quad \textbf{(eq10)}$$

It is widely used to evaluate the performance of hydrological models (Wilcox et al., 1990). It 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. Values for E_{NS} range from negative infinity (poor model) to 1.0 (perfect model). A value of 0.0 means, that the model predictions are just as accurate as using the measured data average. A value greater than 0.0 means, that the model is a better predictor of the measured data than the measured data average. The E_{NS} index is an improvement over R^2 for model evaluation purposes because it is sensitive to differences in the measured and model-estimated means and variance. A major disadvantage of Nash-Sutcliffe is the fact that the differences between the measured and simulated values are calculated as squared values and this places emphasis on peak flows. As a result the impact of larger values in a time series are strongly overestimated whereas lower values are neglected. Values should be above zero to indicate minimally acceptable performance.

Root mean square error – RMSE is determined by calculating the standard deviation of the points from their true position, summing up the measurements, and then taking the square root of the sum. RMSE is used to measure the difference between flow (q) values simulated by a model and actual measured flow (q) values (eq11). Smaller values indicate a better model performance. The range is between 0 (optimal) and infinity.

$$RMSE = \sqrt{\frac{\sum_{i=1}^{n}(q_t^{simulated} - q_t^{measured})^2}{n}} \quad \textbf{(eq11)}$$

Percentage bias – PBIAS (%) measures the average tendency of the simulated flows (q) to be larger or smaller than their observed counter parts (eq12). The optimal value is 0, and positive values indicate a model bias toward underestimation and vice versa.

$$PBIAS = \left(\frac{\sum_{i=1}^{n}(q_t^{measured} - q_t^{simulated})}{\sum_{i=1}^{n}(q_t^{measured})} \right) \cdot 100\% \quad \textbf{(eq12)}$$

The model calibration criteria have been further based on recommended percentages of error for annual water yields suggested from the Montana Department of Environment Quality (2005) who generalised information related to model calibration criteria (Table

4.2). These criteria have been based on a number of research papers (Thormann and Muller, 1982; James and Burges, 1982; Donigian, 1982; ASTM, 1984), and have also been used in Whitehead's (2006) work on East Anglian shallow lakes, where they have been shown to be useful, and gave a good framework for the data comparison during model calibration.

Table 4.2: Model calibration hydrology criteria by Montana Department of Environment Quality (2005)

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

4.3 CALIBRATION

4.3.1 Hydrological Calibration

Hydrological calibration has to be performed to fit the measured daily stream flow data to the simulated daily stream flow. SWAT runs have been performed on a daily time step and output files have been generated on a daily basis. Different model parameters were varied in stepwise fashion, within a reasonable range during numerous calibration runs until final and satisfactory agreement between measured and simulated stream flow was achieved. The calibration was processed entirely at the catchment scale level.

Model base flow predictions are an important part in calibration of the study area hydrology. Total flow is composed of base and surface flow and base flow normally represents the main portion of measured stream flow in the study area. To perform comparison between simulated and measured base flow, estimates of base flow discharge volumes for simulated and measured data, are required. For the separation of base flow the Arnold et al. (1995) method has been used (Automated Base Flow Separation – ABFS) as programmed in the Baseflow Program (BFP, 2007). The

separation has been performed separately for the calibration and validation periods. The same method has been used in the calibration to determine whether SWAT was modelling groundwater correctly. In general, the fraction of water yield contributed by baseflow should fall between the value for Baseflow Fr1 and Baseflow Fr2 as suggested in the Baseflow Program manual (BFP, 2007). As base flow in reality also represents a high proportion of the total flow, values for Baseflow Fr1 were used in further calculations. Figures 4.1 and 4.2 show the separation of base flow from measured mean total flow at the Whitford stream flow gauging station (sub-catchment 10 outlet).

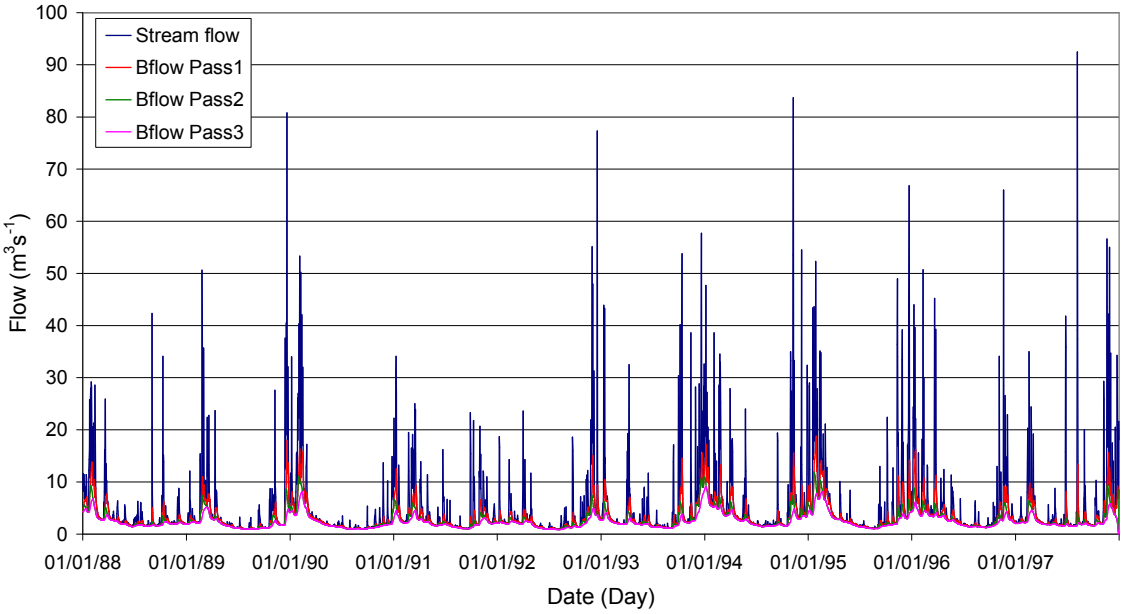


Figure 4.1: Base flow separation for measured data at Whitford using the ABFS method for the calibration period (1988 – 1997).

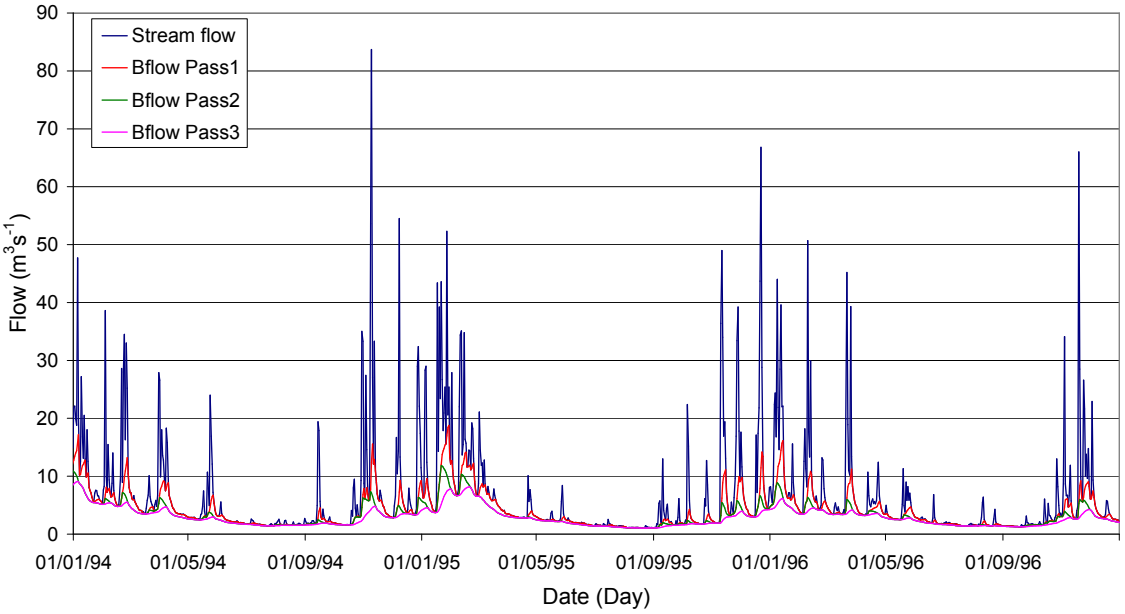


Figure 4.2: Detailed view of base flow separation at Whitford using the ABFS method for the calibration years 1994 and 1995.

During the calibration process SWAT hydrological and nutrient parameters were changed to get better representation of water balance and performance statistics (Table 4.3). The most important and sensitive parameters in the model, were found by others to be groundwater (*.gw), basins (*.bsn) and management (*.mgt) parameters. SWAT parameters commonly used during model calibration are: Base flow alpha factor (ALPHA_BF), Available water capacity (AWC), Soil evaporation compensation factor (ESCO), Groundwater re-evaporation coefficient (GWREVAP), Minimum depth of water in soil for base flow to occur (GWQMN), Groundwater delay time for base flow to occur (GW_DELAY), Minimum depth of water in shallow aquifer for re-evaporation to occur (REVAPMN), Saturated hydraulic conductivity (K_{SAT}), Curve number (CN2).

Table 4.3: Initial and final values of the calibrated hydrological parameters in SWAT model.

Parameter	SWAT Abbreviation	Initial value	Calibrated value
.gw			
Baseflow alpha factor	ALPHA_BF (days)	0.048	0.600
Groundwater re-evaporation coefficient	GW_REWAP	0.020	0.080
Threshold depth of water in shallow aquifer required for return flow to occur	GWQMN (mm H_2O)	0.000	0.650
Groundwater delay time	GW_DELAY (days)	31.000	105.00
.bsn			
Soil evaporation compensation factor	ESCO (mm)	0.950	1.000
Surface runoff lag coefficient	SURLAG (days)	4.000	0.688

4.3.1.1 Annual Flow Summaries

Daily time series for stream flow are only available for the Whitford gauging station at sub-catchment 10 outlet. Annual summary statistics for the calibration period are represented in Table 4.4. Because of the uncertainty involved in determining which crops are grown from year to year, spatial distributions of crop rotations and grazing pastures, soil association distribution and weather characteristics, as well as model inaccuracies, exact agreement was not reached. However the main problem of the hydrological calibration has been the changing flow pattern toward higher flows during the study period from 1986 - 2005. This has affected the total amount of water and water evaporated and hence contribution to stream flow and has had great influence on calibration performance statistics. Generally the model under predicts total flow during the calibration period by 6.56%.

Table 4.4: Annual summary statistics for the calibration period (1988 – 1997) at Whitford; sub-catchment 10 outlet.

Year	Measured flow (mm)	Simulated flow (mm)	Simulated/Measured ratio	Precipitation (mm)	PET (mm)	ET (mm)
1988	504.07	435.51	0.86	930.00	597.15	404.26
1989	501.77	540.91	1.08	948.00	697.05	335.28
1990	474.48	491.60	1.04	881.00	662.73	333.00
1991	496.18	463.56	0.93	929.00	586.21	350.52
1992	440.24	406.52	0.92	874.00	550.74	314.37
1993	611.79	631.02	1.03	1150.00	454.74	309.89
1994	758.23	719.46	0.95	1188.00	545.30	353.27
1995	739.23	717.94	0.97	1088.00	621.64	303.02
1996	625.08	593.01	0.95	949.00	556.56	324.58
1997	669.82	529.14	0.79	1136.00	490.72	335.77
Average	**582.09**	**543.87**	0.93	**1007.30**	**576.28**	**336.40**

Table 4.5 shows the results of measured and simulated base flow separation and Figure 4.3 shows the comparison of measured and simulated base flows at the Whitford gauging station (sub-catchment 10 outlet) for the calibration period.

Table 4.5: Measured and simulated base flow results for the calibration period (1988 – 1997) as calculated by ABFS.

	Base flow as fraction of stream flow		
Type of Data	**Baseflow Pass 1**	**Baseflow Pass 2**	**Baseflow Pass 3**
Measured	0.63	0.51	0.46
Simulated	0.69	0.59	0.54

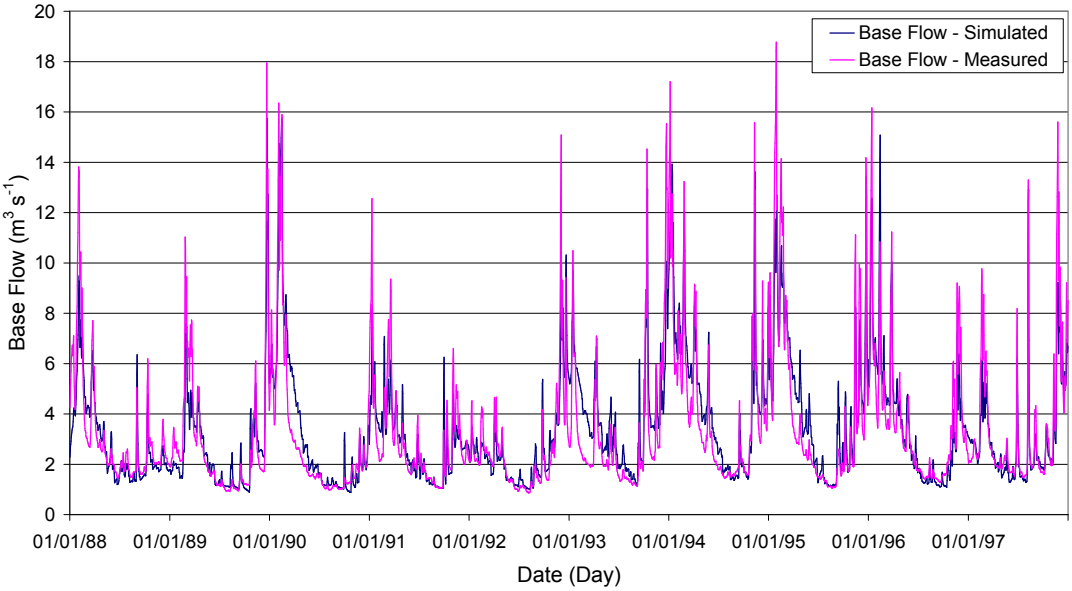


Figure 4.3: Comparison of measured and simulated base flows (Pass 1) at Whitford for the calibration period (1988 – 1997).

From Table 4.6 it can be seen that model simulates the overall annual contribution of groundwater to total flow at Whitford on a yearly level, within the range of acceptability with a 2% over prediction for the calibration period (1988 – 1997).

Table 4.6: Measured and simulated groundwater flow contribution to total stream flow for the calibration period (1988-1997) at Whitford sub-catchment 10 outlet, using ABFS method.

	Groundwater Flow		Total Stream Flow	
Year	**Measured (Separated) (63%) (mm)**	**Simulated (Separated) (69%) (mm)**	**Measured (mm)**	**Simulated (mm)**
1988	317.56	300.50	504.07	435.51
1989	316.12	311.13	501.77	540.91
1990	298.92	339.20	474.48	491.60
1991	312.59	319.86	496.18	463.56
1992	277.35	280.50	440.24	406.52
1993	385.43	435.41	611.79	631.02
1994	477.68	496.43	758.23	719.46
1995	465.71	495.38	739.23	717.94
1996	393.80	409.18	625.08	593.01
1997	421.99	365.11	669.82	529.14
Total	**3667.16**	**3752.68**	**5820.89**	**5438.67**

4.3.1.2 Daily Time Flow Series

After flow calibration the correlation between measured and calibrated simulated daily flow is relatively good as can be seen from Table 4.7.

Table 4.7: Daily time step stream flow model performance statistic for the Whitford gauging station in the river Axe catchment for the calibration period (1988 – 1997).

Statistical test	Units	Optimal values	Base flow	Total flow
R^2	-	1	0.80	0.62
E_{NS}	-	1	0.80	0.62
RMSE	-	0 (lower value - a better model performance)	1.18	4.68
PBIAS	%	0 (+ values = underestimate; - values overestimate)	-2.13	6.57

Performance statistics show that overall simulated stream flow, relative to the measured flow is acceptable, with a tendency to under estimation. Through base flow separation it has been shown that simulated groundwater flow contributes approximately 69% of total flow. The model slightly over predicts base flow (negative PBIAS value), and under predicts total flow (positive PBIAS value). The correlation coefficient (R^2) for daily stream flow is controlled by low flow periods. The value for R^2 (0.62) results from over prediction of lower flows in spring and summer. The lower value for model efficiency (E_{NS}) indicates underestimation of peak flows in autumn and winter. The underestimation of peak flows is also clear from Figure 4.4. The total flow value for R^2 falls into the good category and RMSE value into very good as defined by Henriksen et al. (2003); E_{NS} value falls into the satisfactory category, and PBIAS falls into the very good category as defined by Harmel et al. (2006). In general, the model is able to simulate the temporal dynamics of the river flow. The main problem lays in underestimation of high peak flows in autumn and winter and slow response in flow recession in spring after main rainfall events. Figures 4.4 and 4.5 show visual comparison of simulated and measured river flow.

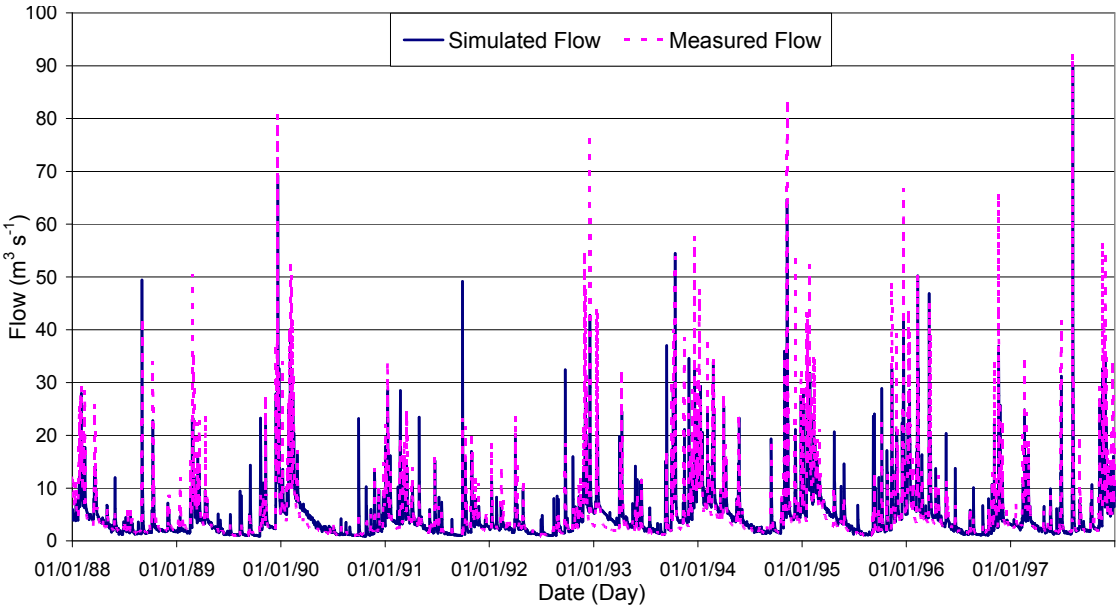


Figure 4.4: Comparison between simulated and measured flow at Whitford for the calibration period (1988-1997).

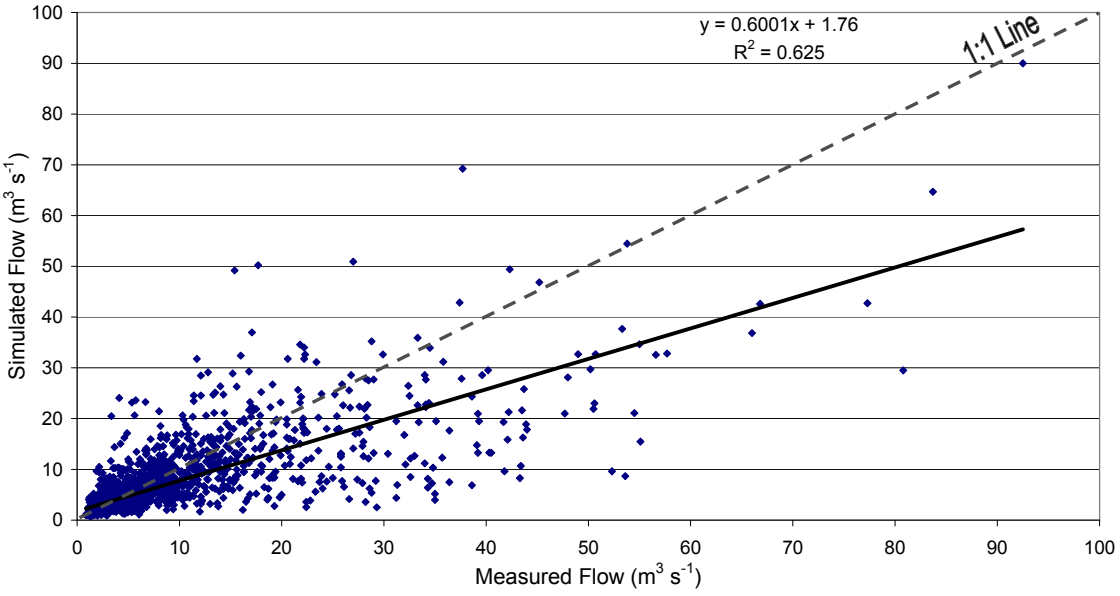


Figure 4.5: Relationship between measured and simulated daily stream flow at Whitford for the calibration period (1988-1997) and corresponding coefficient of determination (R^2)

Table 4.8 shows the acceptable hydrologic calibration criteria in percentage of error for total volume, lowest and highest flows and each season as defined by Montana Department of Environment Quality (2005). Seasons have been defined as: spring (March, April, May), summer (June, July, August), autumn (September, October, November) and winter (December, January, February)

Table 4.8: Acceptable model calibration hydrology criteria for the Whitford gauging station during calibration period (1988-1997).

Errors (Simulated – Observed)	Recommended Criteria (%)	Whitford Statistic (%)
Error in total volume	10	-6.54
Error in 50% of lowest flows	10	-0.53
Error in 10% of highest flows	15	-4.72
Seasonal volume error (spring)	30	5.98
Seasonal volume error (summer)	30	10.42
Seasonal volume error (autumn)	30	-10.45
Seasonal volume error (winter)	30	-14.60

For the calibrated SWAT model the percentage of errors is small or moderate and meets hydrological criteria. Negative values indicate under estimation of total flow and vice-versa. The highest percentage error is occurring in autumn and winter total flow values. In this period of time flows are under predicted and the same tendency can be seen in the 10% of highest flows. On the other hand spring and summer flows are over estimated and this has resulted in low error for 50% lowest flows. Figure 4.6 show the cumulative flow distribution at Whitford.

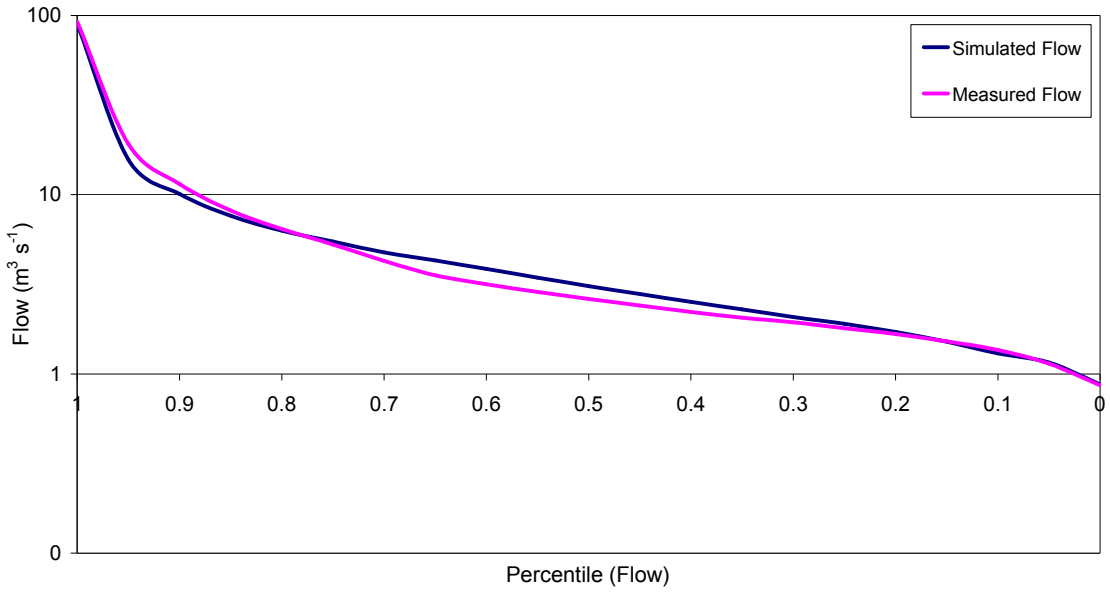


Figure 4.6: Comparison of cumulative distribution of simulated and measured flow on logarithmic scale for the calibration period (1988-1997) at Whitford gauging station.

4.3.2 Model Performance Indicators

An important step before calibrating sediment and water quality parameters is to look at other model performance indicators. Three main parameters are crop growth, evapotranspiration and soil water content, as all of them have a great effect on the water balance. The time series were extracted from the SWAT output files.

Potential evapotranspiration (PET)

Evapotranspiration is a primary mechanism by which water is removed from the catchment. As evapotranspiration is roughly 62 % of the precipitation that falls on the continents 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 (Neitsch et al, 2005).

Simulated potential evapotranspiration (PET) is in agreement with average data reported in Smith (1976). The reported average value (1941-1970) for potential evapotranspiration (PET) is 504 mm and for effective transpiration (ET) 415 mm. The average simulated values for the calibration period (1988-1997) are for PET 576 mm and for ET 336 mm. The average monthly values for the calibration period are in the same range as reported by Smith (1976), although the model slightly under predicts summer values and over predicts winter values. The difference between simulated and reported values could be due to different methods used for calculation of PET and to changes in climate since publication of the data in 1976.

The highest daily values for evapotranspiration were achieved during high temperature months (June, July and August). The highest values varied between years and ranged from 4 mm day^{-1} to a maximum value of 7.77 mm day^{-1} in the dry summer of 1995 (Figure 4.7).

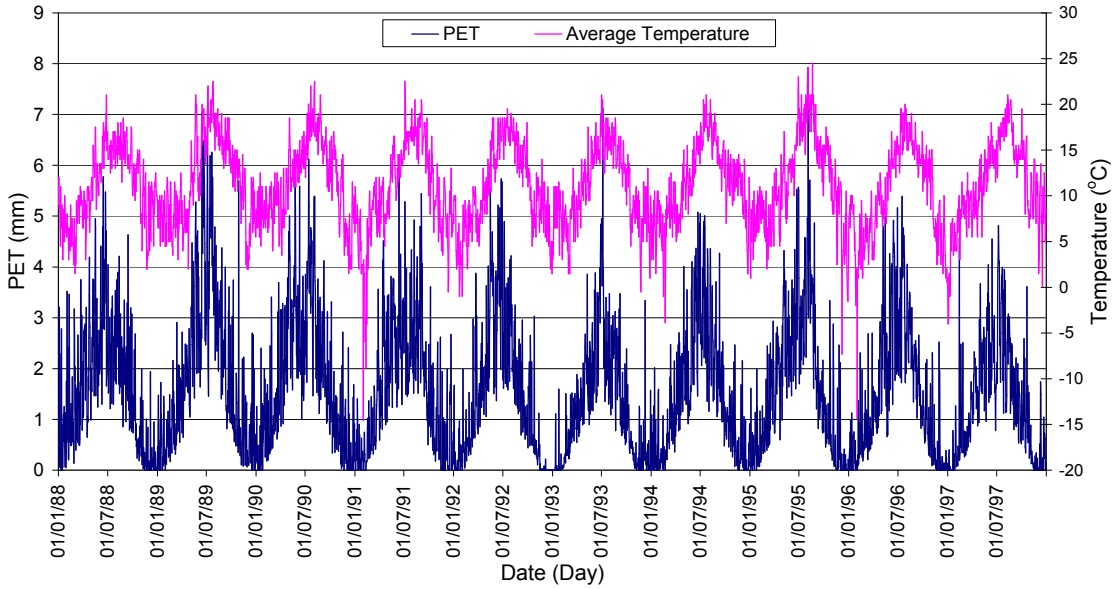


Figure 4.7: Simulated evapotranspiration (Penman-Monteith method, data from Sidmouth) and measured temperature for the calibration period (1988-1997).

Soil water (SW)

Water that enters the soil may move along one of several different pathways. The water may be removed from the soil by plant uptake or evaporation, or it may percolate past the bottom of the soil profile or may move laterally in the profile. Of these three different pathways, plant uptake removes the majority of water that enters the soil profile (Neitsch et al., 2005). The soil water content will be represented correctly if crops and grass are growing at the expected rate and soils have been correctly parameterised. The simulated mean soil water content value, for HRU 57 (Whimple 3 - 572f soil association, PSG4 land management) for the calibration period is 219 mm (Figure 4.8). This soil series is a good representation of actual soil conditions in the catchment as the majority of soils are clayey or silty and the study area is generally wet, with seasonally waterlogged soils.

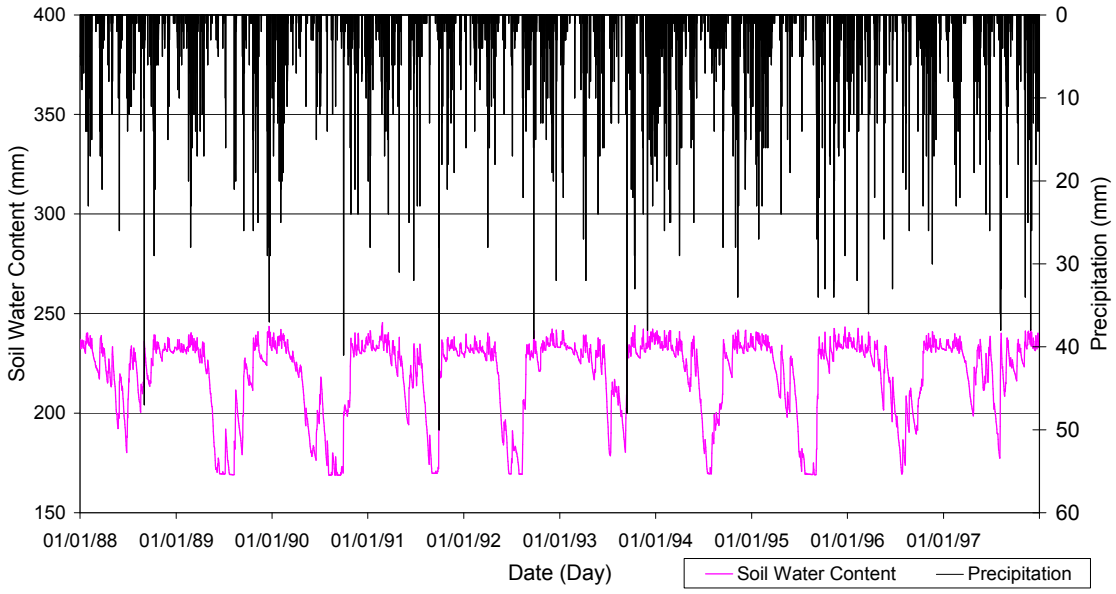


Figure 4.8: Comparison of simulated soil water content (mm) and measured precipitation (mm) for HRU 57 (Whimple 3, PSG4) during calibration period (1988-1997).

The variation in SWC (169 – 245 mm) is also in agreement with mean values for maximum summer soil moisture deficit (98 mm; quartile range 74-111 mm) reported by Smith (1976). The reported date for return to field capacity is 20th October with quartile range between 30th September and 21st November; end of field capacity is 22nd April with quartile range between 31st March and 18th May. Findley et al. (1984) reports that profile available water for grass ranges from 115 to 120 mm between different Whimple 3 soils associations in South West England. Figure 4.9 shows modelled soil water content for the Whimple 3 soil series in sub-catchment 10 for permanent grassland. At permanent wilting point (red line) plant growth will stop (wilting will occur) and it will not recover. This point is never reached within SWAT; therefore none of the plants will die because of water stress. At field capacity (blue line) the soils are filled with water and allowed to drain by gravity until all drainage has ceased. SWAT predicts the end of and return to field capacity between reported ranges as can be observed from Figure 4.9.

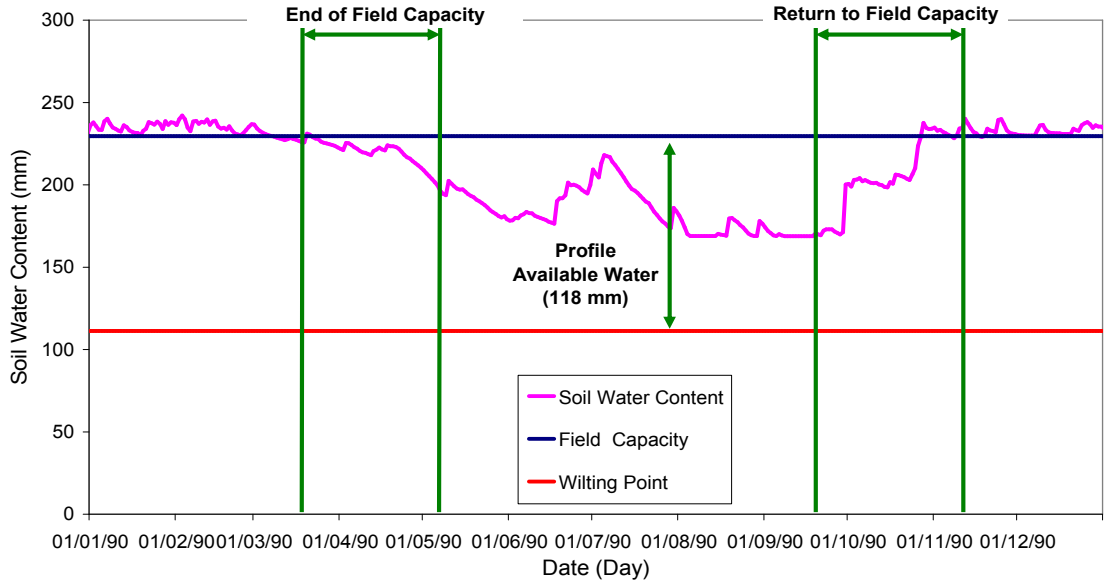


Figure 4.9: Comparison of published soil water content parameters (Findley et al., 1984; Smith, 1976) with simulated soil water content for HRU 57 (Whimple 3, PSG4) for calibration year 1990.

Crop growth

The plant growth component of SWAT is a simplified version of the EPIC (Williams et al. 1984) plant growth model. Phenological plant development is based on daily accumulated heat units, potential biomass is based on a method developed by Monteith, a harvest index is used to calculate yield, and plant growth can be inhibited by temperature, water, nitrogen or phosphorus stress. Plant growth is modelled by simulating leaf area development, light interception and conversion of intercepted light into biomass assuming plant species-specific radiation-use efficiency (Neitsch et al., 2005). In the crop database a range of parameters can be changed to meet requirements for optimal plant growth. Parameters were gathered from various sources (Hough, 1990; Frame, 1992; Kannan, 2003; Whitehead, 2005; HGCA, 2006; Wilkinson et al., 1999; Jones 1988). Plant growth parameters used in this study can be seen in Appendices J1 and J2.

An example crop growth profile for development of leaf area index (LAI) and plant biomass (BIOM) for winter wheat is presented in Figure 4.10; for other crops (winter barley, maize, rye grass) and for permanent grassland growth profiles are given in

Appendices K1 to K4. Peak LAI values for cereals can vary between 3 and 8 and depend on factors such as sowing date, nitrogen fertiliser and water supply (Hough, 1990). All modelled values for LAI lie above values suggested by Hough (1990) and are in reasonable ranges as plants are developing correctly during the modelling period. Perennials (permanent grass) and cold season crops (winter barley, winter wheat, rye grass) go dormant as day length nears the shortest or minimum day length for the year; in that period plants do not grow. As leaf area index starts to decline plants reach senescence. It was observed that the model has some problems with the relationship between senescence and biomass development. As can be observed from Figure 4.10 for winter wheat and from Appendix K2 for winter barley, the model still simulates biomass growth when crops have already reached senescence. At zero leaf area index the plant is totally dry or crop/grass is harvested. Permanent grass biomass never drops to zero and, as can be seen from Appendix K1, harvest operations remove a portion of the biomass and leaves remaining at a site. Biomass production and harvested yields are in the ranges reported by ABC (2006), Hough (1990), Jones (1988) and Wilkinson et al., (1999), although in certain years biomass production is overestimated.

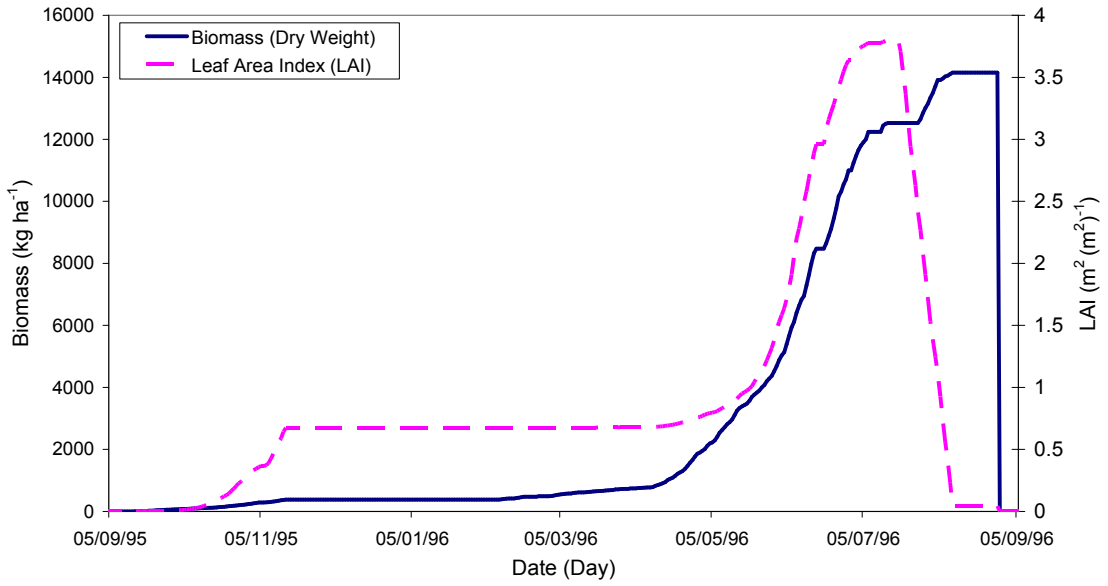


Figure 4.10: Modelled plant growth for winter wheat – ROT1 (Sub-catchment 1, HRU 5).

4.3.3 Sediment Calibration

To calibrate phosphorus the correct representation of sediment yield is required, as phosphorus moves predominantly adsorbed to the sediment particles. As stream flow (surface and base flow contribution) calibration has been performed, the sediment loadings from HRU-s can be calibrated. SWAT models two sources of sediment, from HRU-s and from channel degradation and deposition. SWAT parameters commonly used during model sediment calibration are: USLE crop management factor (USLE_P), USLE slope length (SLSUBBSN), Slope of HRU-s (SLOPE), Crop practice factor (RSDCO), Bio-mixing efficiency (BIOMIX).

The EA do not routinely monitor suspended sediment concentrations and so data were not available for calibration at a daily level. However the EA Action Plan (EA, 2004-a) includes a short report on suspended sediment levels obtained in 2002 for the River Axe Salmon Action Plan. This reports that the highest suspended sediment concentrations can reach more than 1000 mg l^{-1}, and that the mean level of suspended solids at Whitford Bridge in 2002 was 57.7 mg l^{-1}, which is still in excess of the Environmental Quality Standard (EQS) of 25 mg l^{-1}. As time series data for suspended solids were not available, the only comparison that can be done is between reported measured (mean annual value and maximum value for 2002) and simulated values. As sediment loadings are not specifically the subject of this study the exact calibration to mg per litre is not required. However, as a consequence of wrongly calibrated (too high or too low) sediment values the phosphorus values could be simulated inappropriately.

Figure 4.11 shows simulated sediment values at Whitford for the year 2002. The green line represents reported the mean annual value (57.7 mg l^{-1}) obtained from EA, 2004-a, and the pink line represents the simulated mean annual value (60.4 mg l^{-1}) for the year 2002. The action plan reports that maximum values at certain time of the year can exceed 1000 mg l^{-1}, but the model did not simulate values this high. Maximum concentrations are reached in autumn months after large rainfall events, when soils are already saturated and evapotranspiration is low. Due to the shortage of data it is difficult to make any good comparison. As can be seen, the model underestimates the peak

concentrations, which could be related to the fact that the model also underestimates peak flows. It can be observed that sediment concentrations are high throughout much of the year with the highest rates from October to February.

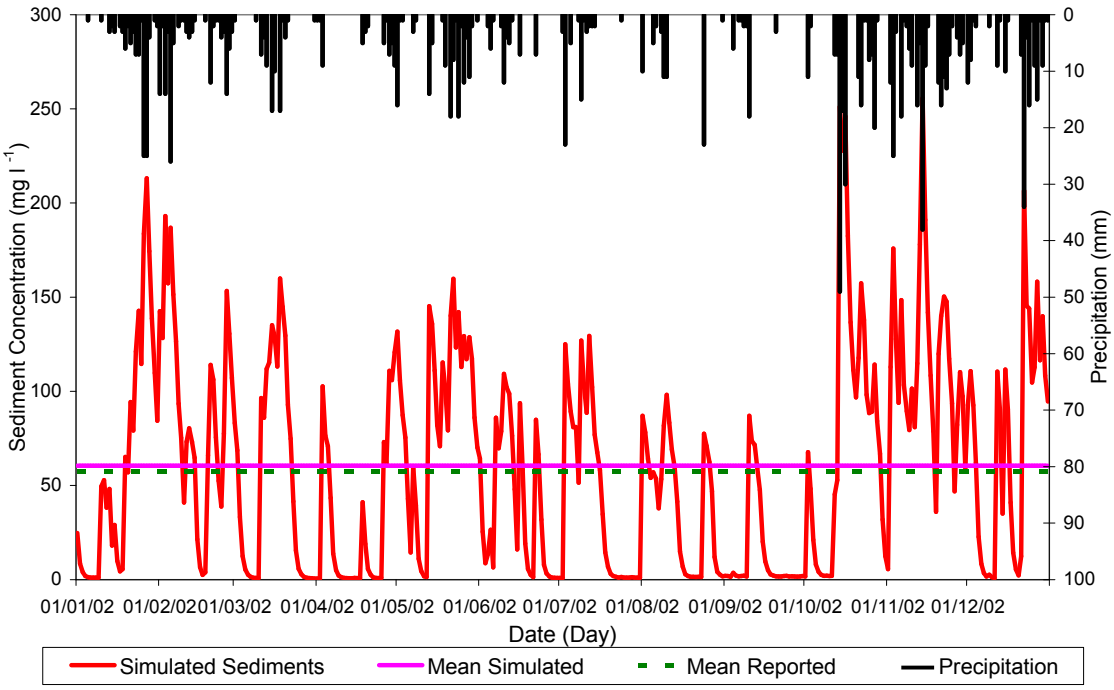


Figure 4.11: Comparison between SWAT simulated sediments (mg l^{-1}) and reported mean sediment concentration (mg l^{-1}) by EA (2004-a) for the year 2002.

4.3.4 Nutrient Calibration

The main four sub-basin output parameters concerning nutrients are nitrate nitrogen (NO3_OUT), organic nitrogen (ORGN_OUT), soluble (mineral) phosphorous (MINP_OUT) and organic phosphorus (ORGP_OUT). Calibration of nitrogen was based on monthly and fortnightly measured nitrate nitrogen (NO_3-N) values (EA) and daily simulated nitrate nitrogen (NO_3-N) values (SWAT). Relationship between nitrate (NO_3^-) and nitrate nitrogen (NO_3-N) is calculated as 1 mg l^{-1} of NO_3-N is 4.4 mg l^{-1} of NO_3^-. On the other hand phosphorus calibration is more complicated. SWAT models two forms of phosphorus (mineral and organic) whilst data supplied by the EA for Whitford gauging station are measured as ortho-phosphorus, which is the major form of mineral phosphorus. Total phosphorus includes three forms: ortho-phosphorus, meta-phosphorus (poly-phosphorus) and organically bound phosphorus (KY Water Watch,

2007). Meta-phosphorus mainly originates in pesticide and is rare. Organically bound phosphorus is simulated by SWAT as organic phosphorus. Therefore, SWAT simulated mineral phosphorus and EA measured values for ortho-phosphorus were assumed to be equivalent for calibration.

SWAT parameters commonly used during model calibration are: Concentration of nitrogen and organic nitrogen in soils (SOL_NO3, SOL_ORGN), Fertiliser application rates (FRT_SURFACE), Crop residue coefficient (RSDOC), Bio-mixing efficiency (BIOMIX), Nitrogen percolation coefficient (NPERCO), Concentration of soluble and organic phosphorus in soils (SOL_MINP, SOL_ORGP), Phosphorus percolation coefficient (PPERCO) and Phosphorus soil partitioning coefficient (PHOSKD). For the purpose of this research only two of them were used, FRT_SURFACE and BIOMIX. Other parameters were not found to be sensitive. The influence on the performance statistics was not great, but still enough for overall improvement. Hydrology parameters can also have a great influence on simulated nitrogen concentration. During this research the GWQMN parameter has been found to play a major role in estimated nitrate concentrations released from groundwater to the river. A very important factor that can change nutrient concentrations in water bodies is fertiliser application rate (FRT_KG – Amount of fertiliser applied to HRU). In the case of too high or too low concentrations in the reach this parameter should be checked first. Table 4.9 represents initial and final values of SWAT parameters used for the nutrient calibration.

Table 4.9: Initial and final values of SWAT model parameters for the calibration of nutrients.

Parameter	SWAT Abbreviation	Initial value	Calibrated value
.mgt			
Fertiliser application rates (Phosphorus applied to arable land use – AGRC (Nitrogen applied at 1st application for maize)	FRT_SURFACE	0.2	0.01
Biological mixing efficiency (Pasture land use - PAUK)	BIOMIX	0.2	0.3

4.3.4.1 Nitrate Calibration

Overall calibration of nitrate nitrogen (NO_3-N) was good; however, variability of the model daily outputs concentrations is high. Major problems were encountered with nitrate concentration (Figure 4.12) in terms of seasonality in certain years or time of the year. The main reason lays in the fact that nitrate is highly mobile and exact agreement is difficult to reach, even more so because the spatial distribution of different crop rotations and management practices, especially harvest and fertiliser application dates and rates are not known. There are also major uncertainties in the soil map, the soil series parameters, the land use map and hence their combination into HRU-s. From the model it can be observed that excess nitrogen mainly comes from certain land uses, which was expected as land use management types were assigned to certain soil types, with specific soil characteristics, that can cause higher losses of nitrogen. All these reasons make concentrations of nitrate highly variable from day to day. It is also important to take into consideration that this is a catchment with a high dairy cow population and with numerous STW. An additional complication is that the Environment Agency takes water samples only on a monthly or fortnightly basis with some limited daily or hourly data. With this mixed sampling frequency is difficult to say if the model is over or under predicting nitrate concentrations, or if the nitrate dynamics are realistic. Johnes (2007) states, that monthly sampling frequency returns imprecise estimates of nutrient loadings for rivers. For visual comparison all measured data were used and compared with simulated daily data.

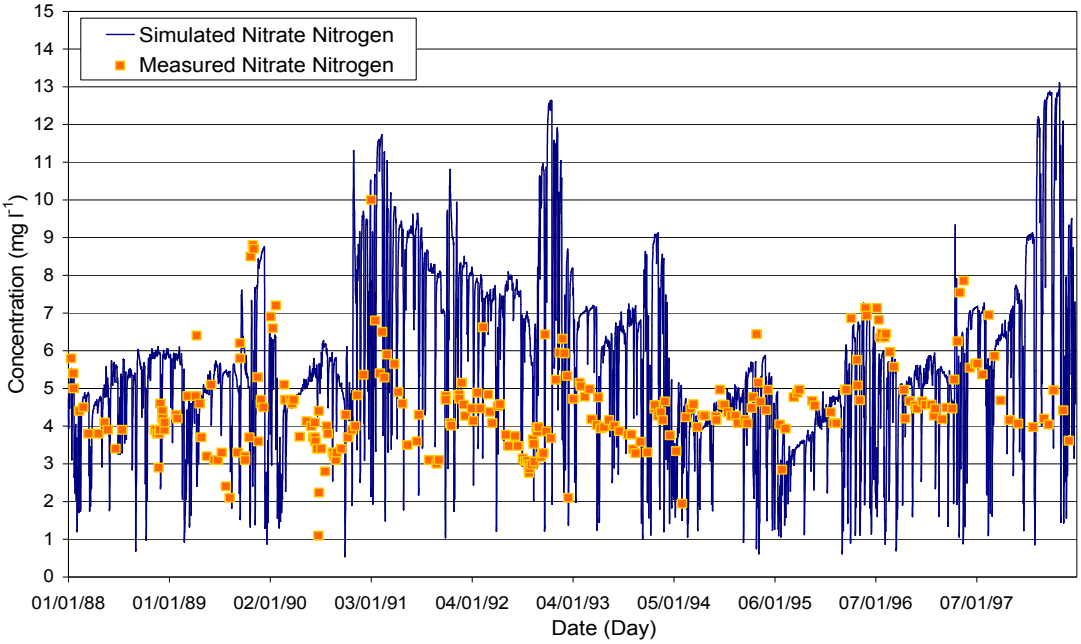


Figure 4.12: Comparison between simulated daily nitrate nitrogen (NO_3-N) concentrations (SWAT) with measured nitrate nitrogen (NO_3-N) concentrations (EA) at Whitford for the calibration period (1988 – 1997).

Problems with high daily variability of nitrate were also observed by Nossent (2005) testing the applicability of SWAT 2005 for modelling water quality in the Grote Nete catchment in Flanders (Belgium). Low nitrate seasonality agreement and high daily variability was also observed by Hutchins et al. (2007) in research on issues of diffuse pollution model complexity arising from performance benchmarking in the Ythan river catchment (UK). They explain the problem of high simulated N peaks, as the dominance of groundwater N contribution. They also noted that increasing spatial representation of soils from lumped to distributed (using the HOST classification), exerted a notable improvement in simulated mean annual N concentration and on qualitative performance. They conclude that sensitivity of individual parameters representing diffuse pollution processes may be greatly diminished by consequences of in-stream processes and mixing of sources, especially point discharges, prevalent in catchments of significant size. Migliaccio et al. (2007) indicates that SWAT algorithms simulating in-stream processes (QUAL2E) still need further testing and refinement, as no significant differences were observed in predictions if in-stream processes were active or inactive.

Figure 4.13 presents a comparison of simulated nitrate nitrogen daily load (kg NO_3-N day^{-1}) and measured nitrate nitrogen loads (kg NO_3-N day^{-1}) recalculated from measured nitrate nitrogen concentrations and mean daily flow. From this comparison better agreement can be observed, however in certain years or times of the year (wetter periods) the model simulates higher loads than were measured. This could indicate that river flow calibration and correct partitioning of water pathways have an important role in water quality modelling. Some of the measured loads are much higher than simulated; this could be explained as a discrepancy between actual flow at the time of water quality sampling and mean flow on the day of sampling, which can be higher or lower.

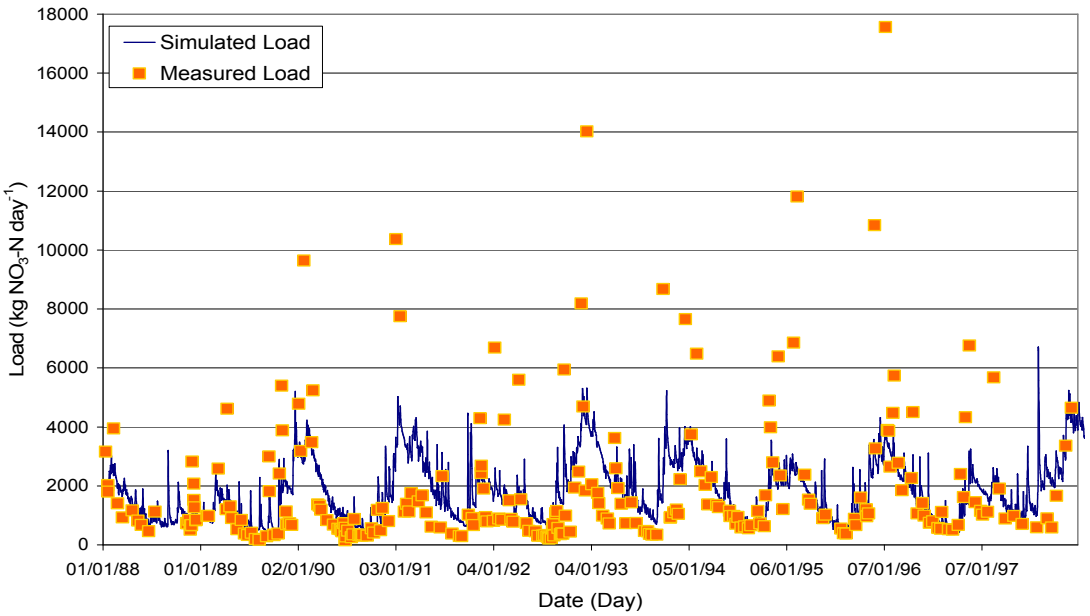


Figure 4.13: Comparison between simulated daily nitrate nitrogen loads (kg N day^{-1}) (SWAT) with measured nitrate nitrogen loads (kg N day^{-1}) (EA) at Whitford for the calibration period (1988 – 1997).

Figures 4.14 and 4.15 represent the cumulative distribution of simulated and measured nitrate nitrogen (NO_3-N) concentration and load, respectively. The cumulative distribution of simulated NO_3-N concentration includes more low and high values and it does not return as many values in the mid-range as measured data. This shows that measured data may not pick up extreme concentration values; this is reasonable as the EA performs the nutrient sampling infrequently. However the cumulative distribution of simulated NO_3-N load includes more low and middle values, and less high values than

measured data. As the majority of measured high loads were recorded in winter this could be a consequence of application of slurry to fields and pastures when soil water content is high. Slurry application cannot be considered in the SWAT model at present. Also, for the recalculation to NO_3-N loads, mean daily flow was used as there were no flow data concurrent with NO_3-N sampling. Because monitored data are available for varying time intervals only the actual days with monitoring were used. Where more than one data point exists for a day these were averaged.

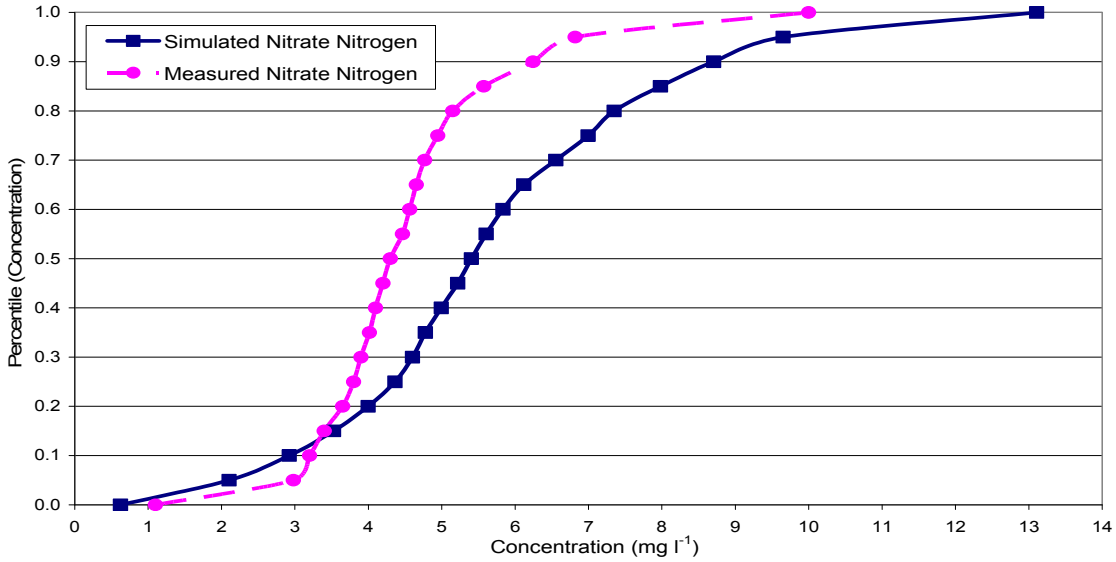


Figure 4.14: Comparison of cumulative distribution of simulated (SWAT) and measured (EA) nitrate nitrogen (NO_3-N) concentrations at Whitford for the calibration period (1988 – 1997).

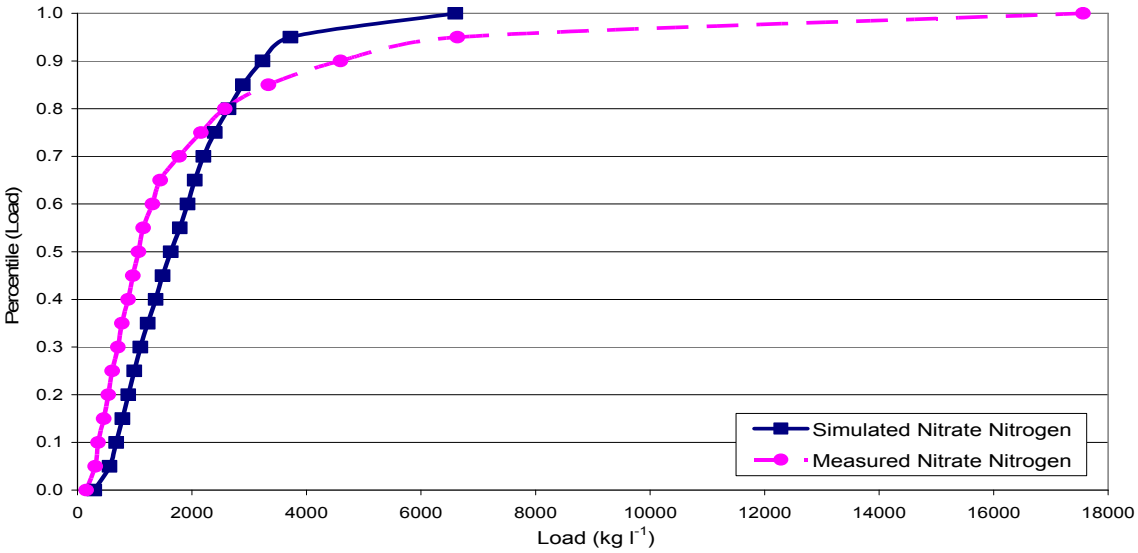


Figure 4.15: Comparison of cumulative distribution of simulated (SWAT) and measured (EA) nitrate nitrogen (NO_3-N) loads at Whitford for the calibration period (1988 – 1997).

As the general picture does not completely explain the behaviour of nitrate nitrogen concentration (high daily variability) in the river it was decided to look in more detail for a period of time when there is 'good' agreement and when there is not. The HRU with the most representative soil type, land use and land management in the study area (HRU 57 - Whimple 3 - 572f soil association, PSG4 - three cut grass silage land management) was selected for this purpose.

The NO_3-N load in groundwater (NO3GW) appears to respond to inputs of NO_3-N from leaching (NO3L) (Figure 4.16). The average fraction of groundwater NO_3-N entering the river reach is high and represents an annual average 92.46 % of total NO_3-N load entering the river in the calibration period. Loads in lateral (NLATQ) and surface flow (NSURQ) are minor in comparison and represent annual averages of 4.51 %, 3.02 %, respectively. It seems that certain soil parameters such as the saturated conductivity (K_{SAT}), which controls routing of water flow and consequentially NO_3-N loads in different flow pathways does not represent the study area well. Various reports (Palmer, 2004; Environment Agency, 2004-a; Environment Agency, 2004-b; Environment Agency, 2003) from the area identify soil compaction originating from heavy machinery and overstocking as the cause of restricted infiltration and hydraulic conductivity, and consequently changed flow pathways.

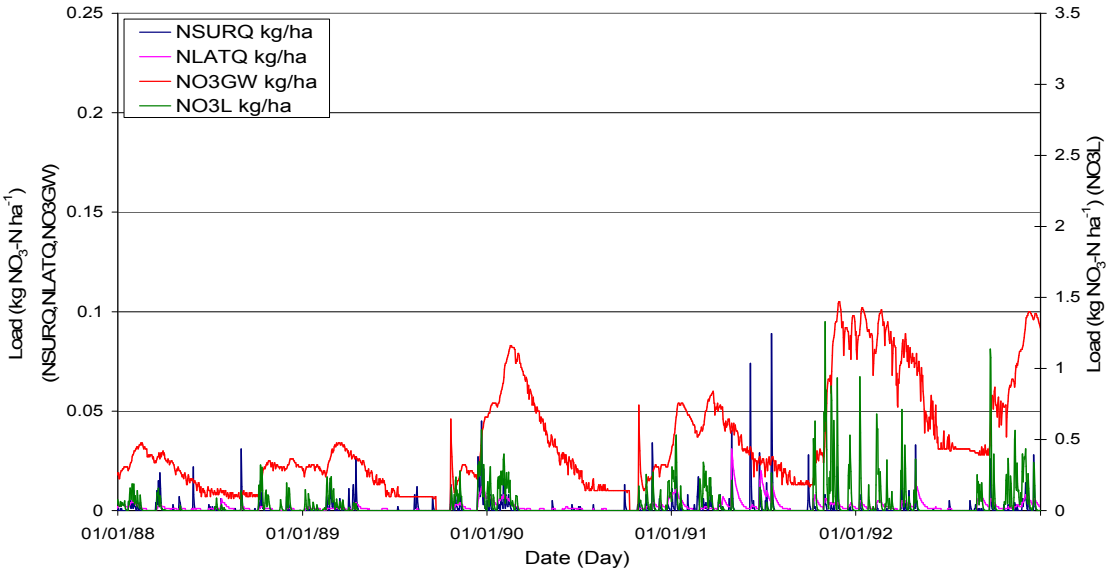


Figure 4.16: Comparison of HRU output for nitrate loads in surface, lateral and groundwater flow and leaching (kg N ha^{-1}) for the calibration years from 1988 to 1992 in HRU 57.

The NO_3-N in groundwater (NO3GW) is very noisy and responds to pulsed inputs from leaching at some times of year (Figure 4.17).

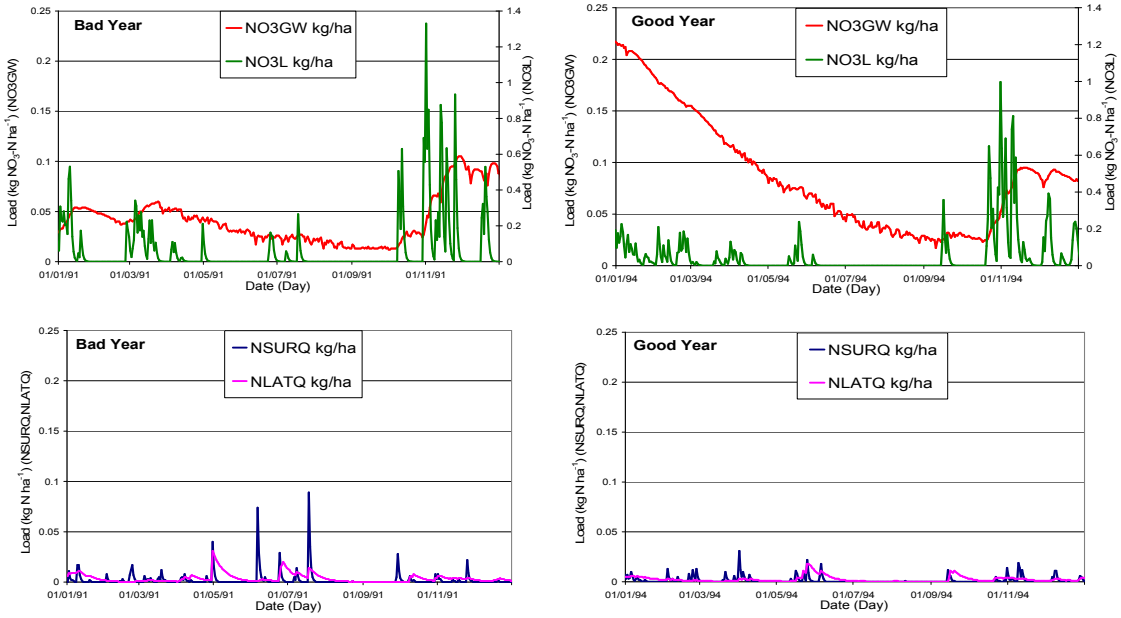


Figure 4.17: Comparison of HRU 57 output for nitrate nitrogen (NO_3-N) in surface, lateral and groundwater flow and leaching (kg N ha^{-1}) for bad and good agreement year.

Figure 4.18 shows that even in the "good" year, dilution effects of higher river flows are too large. The high drops in NO_3-N concentration are the consequence of the high fraction of total NO_3-N load in groundwater flow as compared to that in the river. It seems that there should be more NO_3-N movement simulated also through lateral (majority) and surface flow, to limit this high daily variability.

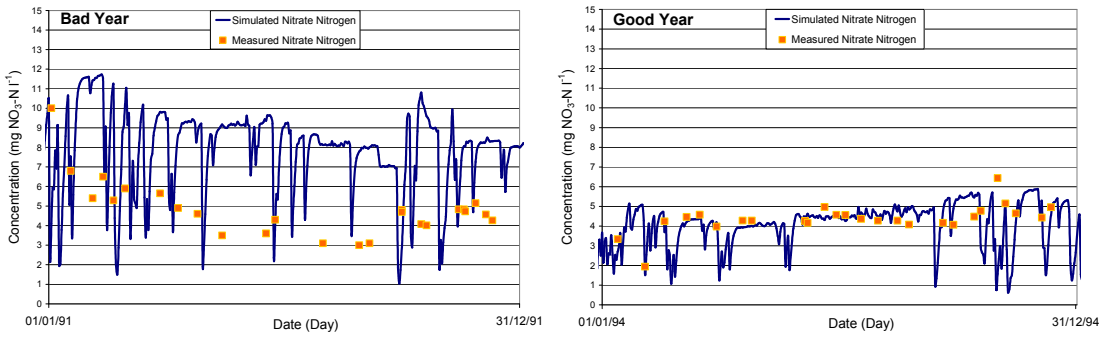


Figure 4.18: Comparison of measured and simulated nitrate nitrogen concentrations (mg NO_3-N l^{-1}) at the Whitford sampling point for bad and good agreement year.

Simulated NO_3-N recession in summer time is very slow as can be seen from Figure 4.19. It was also observed that in certain years NO_3-N concentration in groundwater is just cut off (falls to zero) (Figure 4.16), for example in 1989 and 1990. Nitrate nitrogen loads in the river follow the pattern of groundwater contributions to the river, dropping during the summer when fields move away from field capacity, and increasing dramatically in autumn as soils wet up again (Figure 4.20).

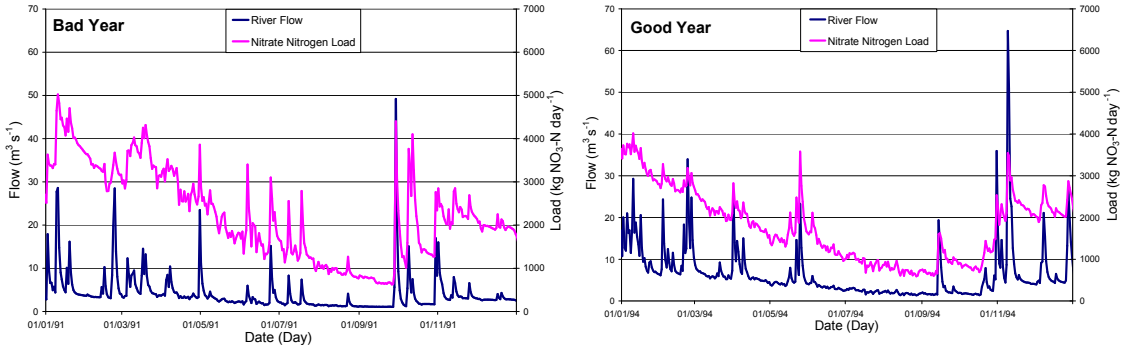


Figure 4.19: River flow (m^3 s^{-1}) and load (kg N) of nitrate transported with water out of reach during time step for good and bad agreement year.

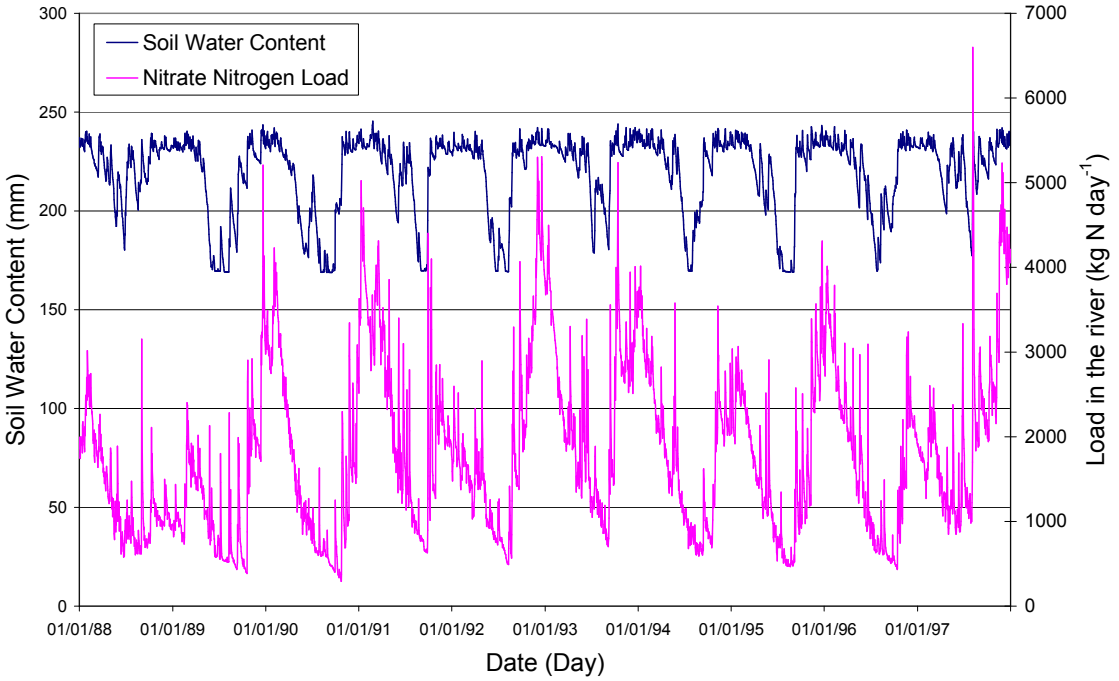


Figure 4.20: River flow (m^3 s^{-1}) and load of nitrate nitrogen transported with water out of reach during time step (kg) for calibration period and soil water content of the representative HRU 57.

During the soil data analysis it was noticed that some of the soil associations, obtained from the national soil map, had high or moderately low soil saturated conductivity

(K_{SAT}) values. This could result in inappropriate simulation of flow pathways. Changes to the NPERCO parameter were also tested to move more nitrogen by surface flow, but due to the fact that the model moves the majority of the water through groundwater or lateral flow, this did not result in better performance of NO_3-N concentrations. SWAT models NO_3-N concentration in water for a soil layer as a function of water available in that layer and nitrogen present. Due to reasons mentioned previously the model has high nitrogen remaining in the soil. Therefore, if the amount of water percolating downwards is high it will leach more nitrogen.

As described in the literature review, research carried out by McMahon and Bohlke (1996) showed that nitrate (NO_3^-) load in discharging groundwater was 15 - 30% lower than the load that would have resulted from the discharge of unaltered groundwater from the recharge area, due to the significant denitrification at locations where oxygen is absent or present in very low concentrations in aquifers with silt and clay confining beds and riparian areas near to stream. According to the SWAT 2005 Input/output documentation and in the Theoretical documentation, this could be simulated via the groundwater parameter called Half-life of nitrate in the shallow aquifer (HLIFE_NGW). However this parameter did not appear to exist in the latest version of SWAT 2005 for ArcView obtained from the official SWAT website. This parameter may help to solve the problems seen in years with poor measured-simulated correlation. Another reason why NO_3-N concentration results do not show seasonality could also be that the SWAT model does not take into account macrophytes (important feature of the river Axe catchment) in the river channel or on the river banks which could reduce nitrogen concentration in the river water during the summer. Due to the limitations of data it was not possible to locate and simulate wetlands which are also an important sink of nitrogen. Wetlands represent only a small part of the study area, but they could have significant influence on nitrogen concentrations.

4.3.4.2 Phosphorus Calibration

Ash et al (2005) suggest that almost 50% of the ortho-phosphorus in the study area enters the river from point discharges, therefore additional attention was given to point source parameters. As there was only a small amount of data available for estimation of nutrient concentrations in point discharges, the results for the calibration period are very encouraging. Figure 4.21 shows the comparison between simulated mineral phosphorus daily concentrations (mg l^{-1}) and measured monthly or fortnightly ortho-phosphorus concentrations (mg l^{-1}) for the calibration period at Whitford gauging station.

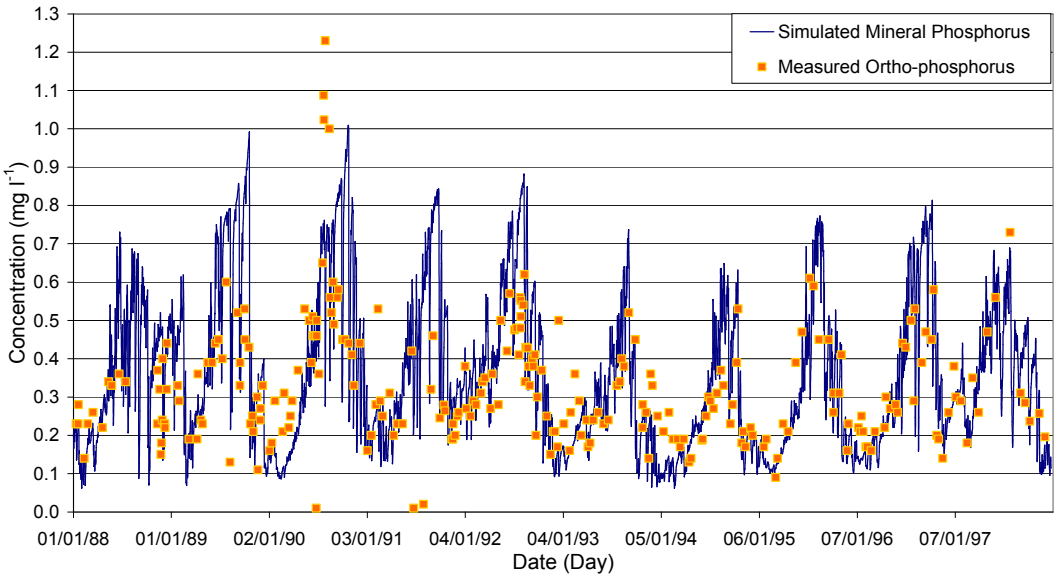


Figure 4.21: Comparison between simulated daily mineral phosphorus (MINP) concentrations (SWAT) with measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) at Whitford for the calibration period (1988 – 1997).

Figure 4.22 represents the cumulative distribution of simulated and measured phosphorus (PO_4^{3-}) concentration. From the figure it can be observed that the simulated data is giving us lower low values and higher high values, but not as many peak values as measured data. This is mainly due to the fact that flow from STW discharges were simulated as constant daily loadings. That means that periods of time when discharge concentrations of phosphorus in outflow were high or low are not modelled correctly.

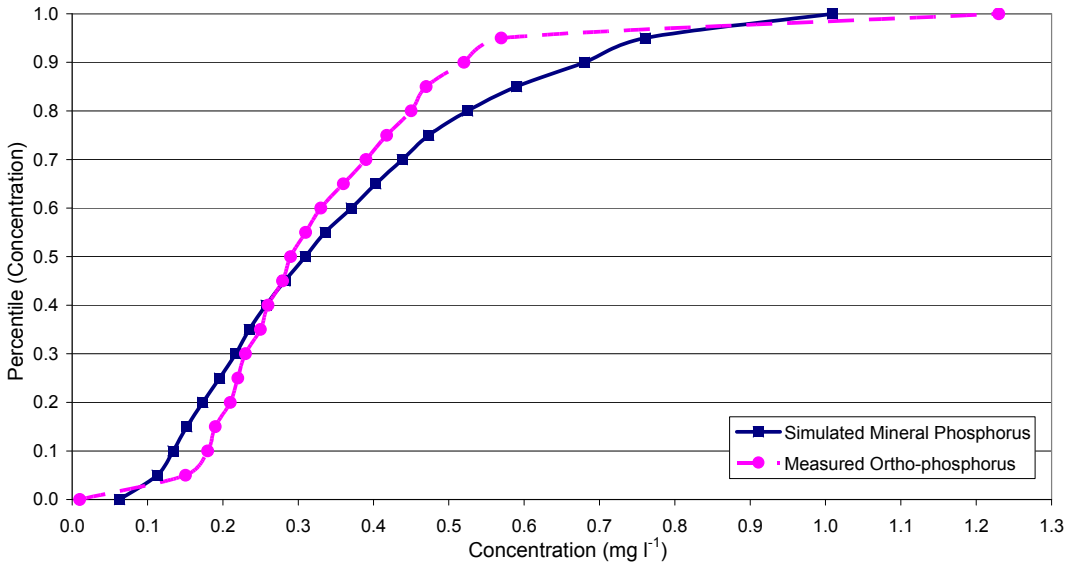


Figure 4.22: Comparison of cumulative distribution of simulated mineral phosphorus (SWAT) concentrations and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) at Whitford for the calibration period (1988 – 1997).

4.4 VALIDATION

The model was validated for the period of 1st January 1998 to 31st December 2005. For this purpose daily measured data from the gauging station at Whitford were used. Results and statistical performance of the model hydrological validation are presented in Table 4.10 and a visual comparison is given in Figure 4.23. The main problem concerning hydrological validation was related to the higher measured flow variance during the validation period. During the study period the catchment became wetter with higher peak flows and with higher base flow. The average flow during the validation period is higher than that for the calibration period by more than 1 $m^3 s^{-1}$. Due to this the performance statistics for the validation period are lower, especially the Nash-Sutcliffe Index (E_{NS}) has returned much lower values (0.47) than in the calibration period (0.62). If the variance had been the same during validation and calibration periods, the Nash-Sutcliffe efficiency coefficient (E_{NS}), for the validation period would be 0.53. Other parameters meet recommended statistical criteria for total flow comparison between simulated and measured flows. Values for R^2 (0.53) and RMSE (7.24) are in the good category (Henriksen et al., 2003). Positive value for PBIAS (15.11; 3.88) indicate

underestimation of total and base flow, but the PBIAS total flow performance statistic is still in the satisfactory category defined by Harmel et al. (2006).

Table 4.10: Daily time step stream flow model performance statistics for the Whitford gauging station in the river Axe catchment for the validation period (1998 – 2005).

Statistical test	Units	Optimal values	Base flow	Total flow
R^2	-	1	0.75	0.53
E_{NS}	-	1	0.74	0.47
RMSE	-	0 (lower value - a better model performance)	1.36	7.24
PBIAS	%	0 (+ values = underestimate; - values overestimate)	3.88	15.11

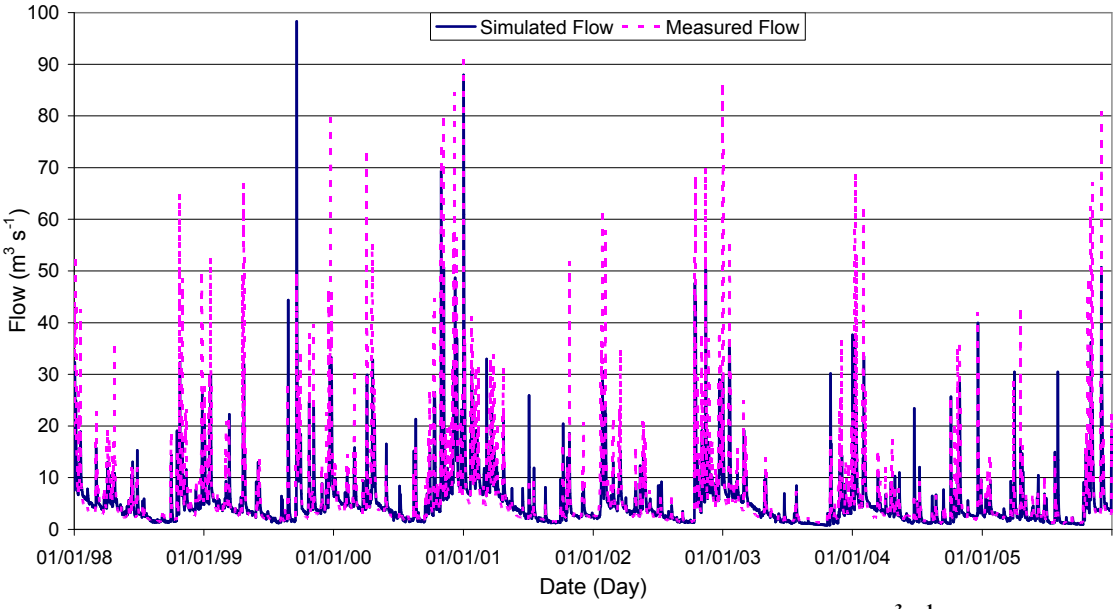


Figure 4.23: Comparison between simulated and measured daily flow values (m^3 s^{-1}) for the Whitford gauging station for the validation period (1998 – 2005).

Sediment validation for the study area could not be performed, as only mean annual concentration and maximum concentration were reported for the year 2002 which were used for calibration.

The nutrient validation has been performed for Whitford gauging station using Environment Agency measured data for nitrate nitrogen (NO_3-N) concentration (Figure 4.24) and load (Figure 4.25) and ortho-phosphorus concentration (Figure 4.27, 4.28).

The data time series were at the same resolution as for the calibration period. Results for NO_3-N (Figure 4.24, 4.25) and phosphorus (Figure 4.27, 4.28) show similar patterns as in the calibration period, but with some major differences. In the validation period peak measured concentrations for NO_3-N and phosphorus are lower; but on the other hand simulated values return higher peak concentrations. The most probable reasons for higher NO_3-N concentrations and high daily variability lie in the reasons already described in calibration. However, in the last five years the EA and different catchment initiatives made great efforts to reduce diffuse and point source nutrients in the study catchment. The main catchment initiative was the Cycleau Project (2004), where the main focus was on diffuse nutrient pollution from farms. The project also focused on unrestricted livestock access to the river, increased maize cultivation, soil erosion and waste management. In last few years a Nitrate Vulnerable Zone (NVZ) action plan was initiated in the lower catchment with the aim of reducing nitrate from agriculture.

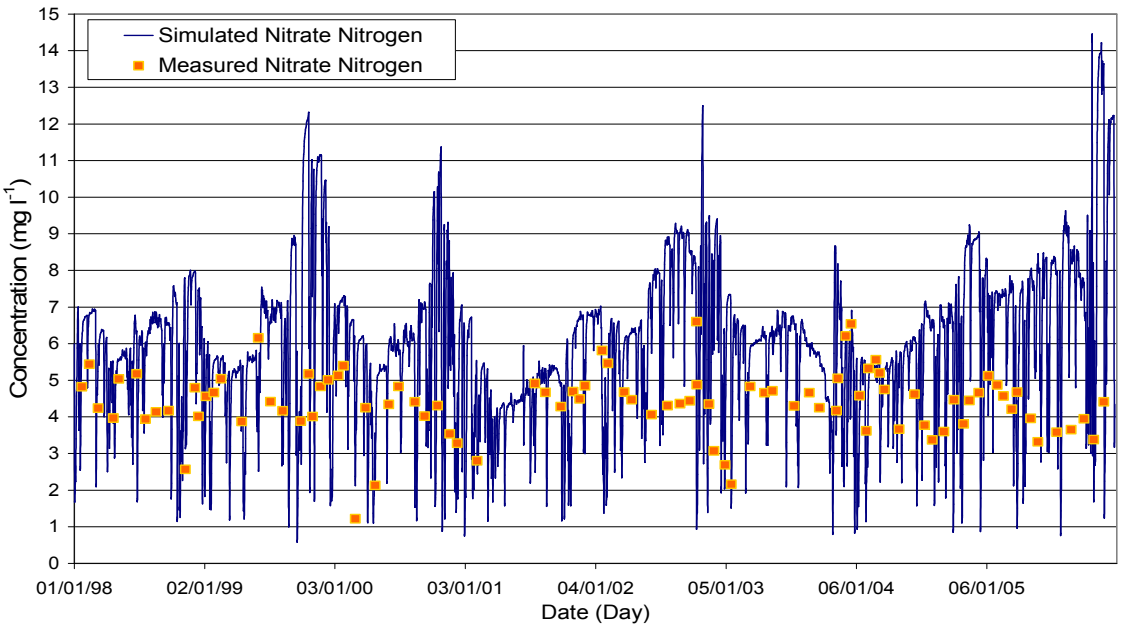


Figure 4.24: Comparison between daily simulated (SWAT) and measured (EA) concentrations of NO_3-N for the gauging station at Whitford during the validation period (1998 – 2005).

Figure 4.26 represents the cumulative distribution of simulated and measured nitrate (NO_3^{-1}) concentration. It can be clearly seen that the simulated data (SWAT) includes more low and high values than the measured data (EA). On the other hand simulated data does not return as many values in the middle range as measured data. From this it could be concluded that measured data does not pick up extreme values; this is

reasonable as EA performs nutrient samplings infrequently. As mentioned previously, problems also lie in definition of soil series layer parameters which are crucial for simulation of water flow routing.

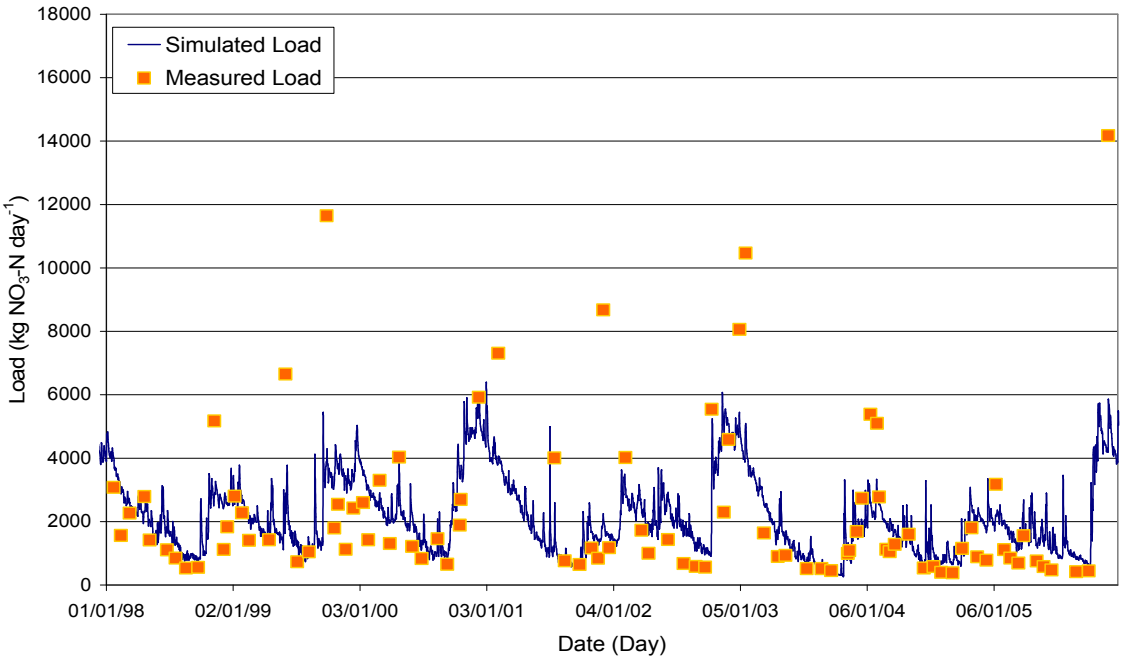


Figure 4.25: Comparison between simulated daily nitrate nitrogen (NO_3-N) loads (SWAT) with measured nitrate nitrogen (NO_3-N) loads (EA) at Whitford for the validation period (1998 – 2005)

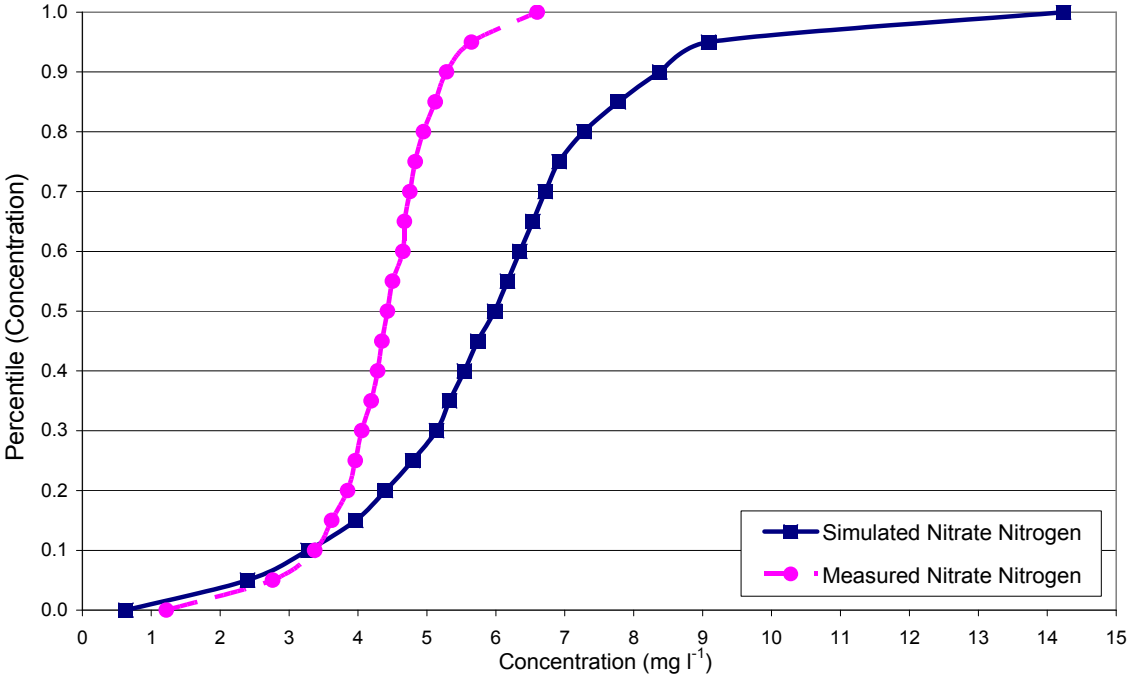


Figure 4.26: Comparison of cumulative distribution of simulated (SWAT) and measured (EA) nitrate nitrogen (NO_3-N) concentrations at Whitford for the validation period (1998 – 2005).

Figure 4.27 shows a large discrepancy between simulated and measured concentrations from the year 2002 onwards, as measured ortho-phosphorus (PO_4^{3-}) concentrations start to decrease. This is due to the fact that phosphorus stripping from discharges was introduced for the main contributors to total load, especially for the creamery (St Ivel Ltd.). To test if lower phosphorus concentrations in discharges do have an impact on mineral phosphorus concentration in the river, the model was run with lower concentrations in outflow from the main 7 point source contributors (four public STW and three industrial plants). Concentrations for public STW were lowered to 2 mg l^{-1} and for industrial sites (creameries) an 86 % reduction from 36 mg l^{-1} to 5 mg l^{-1} was applied. The organic phosphorus concentrations in discharge outflows were lowered proportionately. New values for point source discharges are presented in Appendix I3. The results of this simulation run (Figure 4.28) were encouraging, after lowering PO_4^{3-} concentrations for the major point discharges the daily simulated concentrations for mineral phosphorus fall in to the range of measured concentrations (EA) between 2002 and 2005. As possible land management scenarios to reduce diffuse source nutrients are to be modelled next, these lower values for phosphorus concentrations in discharge effluent were maintained.

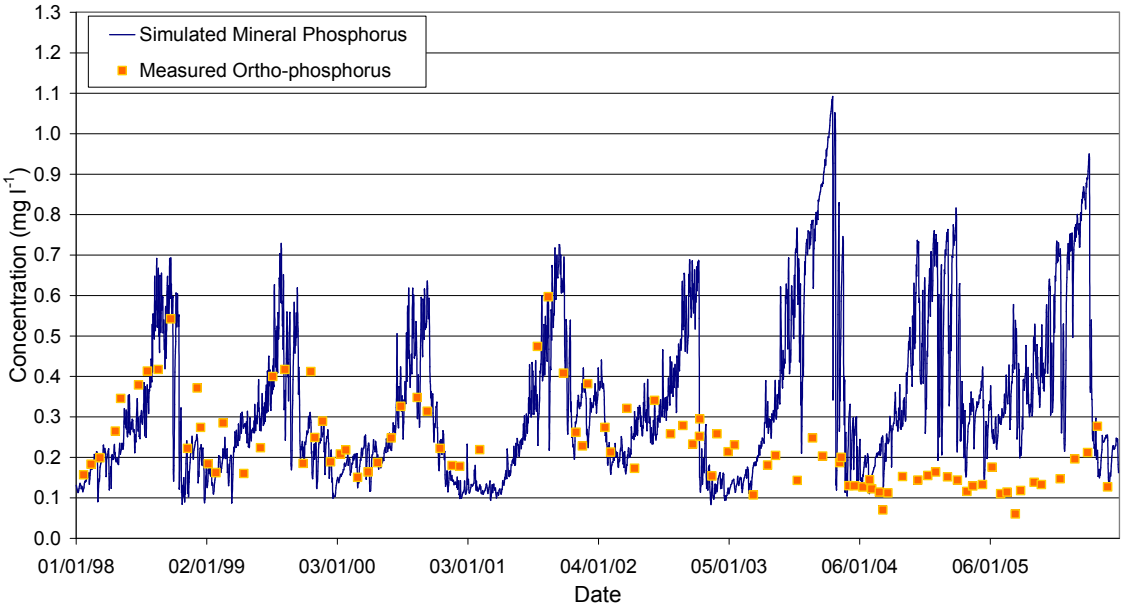


Figure 4.27: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the gauging station at Whitford during the validation period (1998 – 2005) without phosphorus stripping introduction.

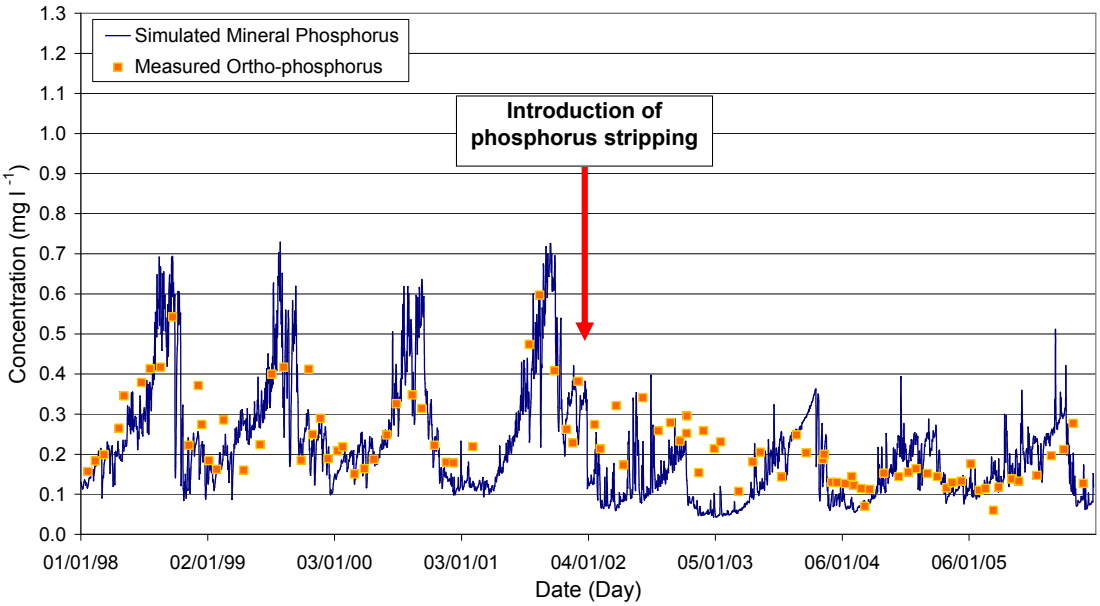


Figure 4.28: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the gauging station at Whitford (1998 – 2005) with phosphorus stripping introduction.

From Figure 4.29 for phosphorus comparison it can be observed that simulated data shows more peak values than measured. The lack of monitoring during extreme events is likely to have an even greater impact for high ortho-phosphorus concentrations. The simulated data (with phosphorus stripping) shows more peak and less low values. This is mainly due to the fact that flow from STW discharges were simulated as constant daily loadings. That means that periods of time when discharge concentrations of phosphorus in outflow were high or low are not modelled correctly.

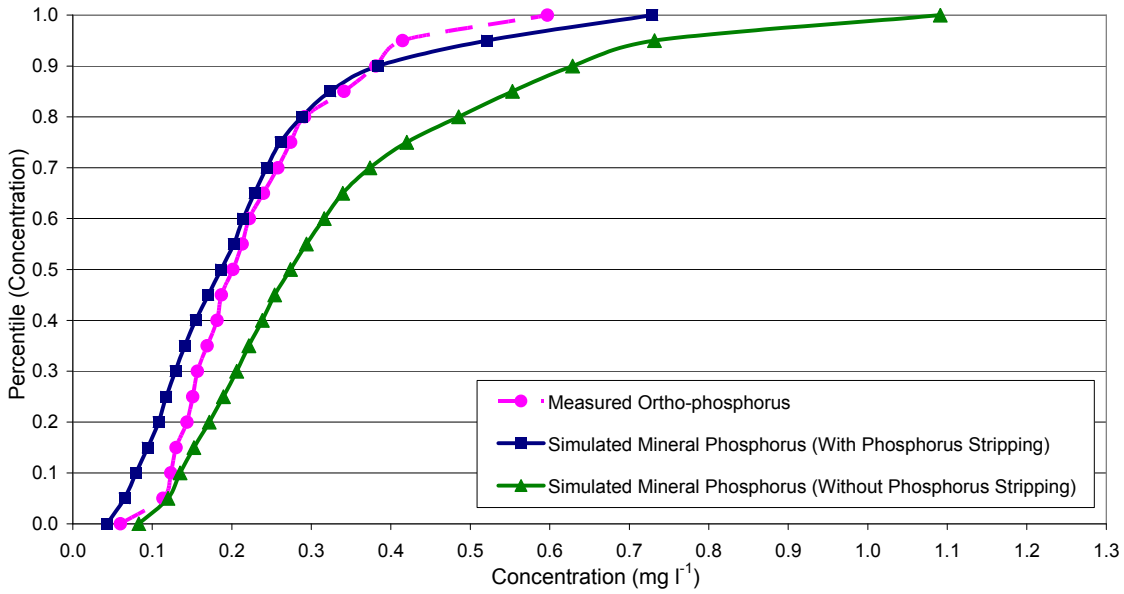


Figure 4.29: Comparison of cumulative distribution of simulated mineral phosphorus (SWAT) concentrations (with and without phosphorus stripping introduction) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) at Whitford for the validation period (1998 – 2005.

Validation of nitrate and phosphorus concentrations was also performed for the nutrient sampling points at the four catchment sections (Table 4.11). Validation results show similar patterns as for sub-catchment 10 outlet (Whitford), with exception of low ortho-phosphorus concentrations in River Yarty section and high nitrate values in River Coly section. Comparison between daily simulated and measured concentrations of nutrients for validation period can be observed in Appendices L1 to L8.

Table 4.11: Comparison of measured and modelled validation results for mean annual concentration of nitrate nitrogen and ortho-phosphorus for four catchment sections.

Catchment section (sampling point)	Mean annual concentration											
	Nitrate Nitrogen (mg NO_3-N l^{-1}) (1998-2005)						Ortho-phosphorus (mg l^{-1}) (1998-2005)					
	measured			simulated			measured			simulated		
	mean	min	max	mean	min	max	mean	min	max	mean	min	max
Sub-catchment 10 (Whitford)	4.38	1.22	6.60	5.94	0.63	14.24	0.23	0.06	0.60	0.21	0.04	0.73
River Yarty (Newenham Abbey)	3.55	2.29	6.76	4.06	0.61	9.41	0.12	0.02	0.33	0.03	0.00	0.65
River Coly (Colyford)	3.03	1.62	5.40	6.56	0.76	23.28	0.10	0.03	0.34	0.06	0.01	0.77
Upper Axe (Slymlakes)	4.64	2.62	6.53	6.25	0.51	17.66	0.26	0.09	2.01	0.24	0.04	0.93
Lower Axe (Axe Bridge)	4.54	1.66	8.49	6.03	0.69	14.05	0.29	0.08	0.90	0.20	0.04	0.68

5 LAND MANAGEMENT SCENARIOS

Land management scenarios are defined to investigate possible different land use management impacts on the behaviour of diffuse source nutrients (nitrate, phosphorus), all with the purpose of delivering water quality improvements in the River Axe catchment. To achieve the aim of this research three different land use management scenarios were applied to the study area. The constructed scenarios include land use management change and best management practices. The final objective of this research is to investigate impacts of defined scenarios on diffuse nutrient reduction and to recommend possible mitigation measures.

Scenarios were developed based firstly on interviews with farmers and Environment Agency representatives for the Axe catchment and further on the basis of site investigation during a field trip to the study area in February 2007. Secondly a range of different reports and action plans produced by Department for Environment, Food and Rural Affairs (DEFRA) - Catchment Sensitive Farming (DEFRA, 2002-a; DEFRA, 2006) and Environment Agency (Environment Agency, 2004-a; Environment Agency, 2004-b; Ash et al., 2005) dealing with environmental problems in the river Axe catchment were taken into consideration. Two of the alternative future scenarios (Extensive and Sheep) are also correlated with the UK Foresight Programme which is linked with future Agricultural Policy Scenarios, or so called future socio-economic and land use scenarios (Morris, 2003). With this information three of the most viable or realistic scenarios which could be introduced and applicable for the study area were constructed. Scenarios were designed to maintain the characteristic landscape of the river Axe as an important feature of European importance. Therefore no extreme features were introduced in to the scenarios, such as afforestation of the study area. Afforestation could reduce nutrient loads to the river, but it would also reduce ecological value, quality of life and economic value especially in respect of tourism development in the study area. Detailed description of the developed scenarios follows. It is difficult to say if scenarios are realistic to the point that farmers would accept them with out hesitation, as they would have to change their way of land management and even livestock production. All changes, even field buffer strips would need to be linked

with educational courses and financial support. With this in mind certain parts of the scenarios, which are easily achievable, could be linked with agricultural subsidy payments at national UK or EU level.

The SWAT parameters presented in Table 5.1 used in the baseline management files were also used for the land management scenarios.

Table 5.1: SWAT parameters used for baseline and new scenarios.

SWAT Parameter	SWAT Symbol	Used Value
Fertiliser application rates (Phosphorus applied to arable land use – AGRC (Nitrogen applied at 1st application for maize)	FRT_SURFACE	0.01
Biological mixing efficiency (Pasture land use – PAUK and PG1S)	BIOMIX	0.3
Minimum plant biomass for grazing (kg/ha) (Pasture land use – PAUK and PG1S)	BIO_MIN	200
Harvest index override (hay and silage grass, maize silage)	HI_OVR	0.95

5.1 SCENARIO 1 – FIELD BUFFER STRIPS

The field buffer strips scenario (FBS) is a function of how to minimize influences of diffuse pollution resulting from agricultural activities, especially from the perspective of soil compaction and over fertilisation without drastic changes in crop or pasture management. For this purpose a range of Best Management Practices (BMP-s) would be implemented.

Field buffer strips reduce transport of pollutants and are considered structural. They are defined as planted or indigenous bands of vegetation that are situated between pollutant source areas and receiving waters to remove pollutants from surface and subsurface runoff (Mostaghimi, 2000). Haycock and Burt (1993-b) suggest that buffer zones encourage slow percolation and long retention times, essential for nitrate removal by assimilation and denitrification. Other important mechanisms are reduction of surface runoff velocities by vegetation roughness and the physical filtering effect of dense vegetation (Muscutt et al., 1993). The effectiveness of buffer zones is closely correlated with their slope and width. Higher slopes are less effective as buffer zones, whilst wider

buffers are more effective. Robinson et al. (1996) observed that 3.0 m wide buffer strips effectively removed up to 70% of the sediment load from cropland runoff. Dillah et al. (1989) observed sediment trapping efficiency for 4.6 m wide buffer strips of 53% to 86%. Buffer zones are expected to function for up to 10 years, but if there is no proper maintenance (regular harvesting of the biomass grown) they may become a source of pollution because of the nutrient accumulation within them.

Field buffer strip structures are also considered to be important habitat within the river corridor, especially if fields are surrounded with hedges. DEFRA's (2007-c) Entry Level Stewardship handbook offers farmers different options, such as which width of filter strip to choose. Farmers can choose between 2 m, 4 m and 6 m buffers strips on cultivated (arable) land and on grassland. Using guidance from this handbook, an option of 4 m buffer strips was chosen to be modelled.

Field buffer strips can be modelled in SWAT through the input data file at the HRU level in management (*.mgt) data file through a parameter called width of edge-of-field filter strip parameter (FILTERW) (Table 5.2).

Table 5.2: SWAT parameters for modelling riparian zones and buffer strips.

Input file	File Level	SWAT Parameter	Description	Baseline value	FBS Scenario value
Field Buffer strips					
Management (*.mgt)	HRU	FILTERW	Width of edge-of-field filter strip (m)	0.00	4.00

For scenario 1 only field buffer strips were added to the each of the arable (AGRC) and pasture (PAUK) HRU-s. All other parameters (fertiliser application rates, dates, tillage operations) in management files (*.mgt) stayed at the same level as for the baseline scenario.

The width of edge-of-field filter strip parameter (FILTERW) reduces the loads of sediment, nutrient, pesticide and bacteria as the surface runoff passes through the filter strip. Nutrient loads in surface and subsurface flow are trapped in strip vegetation. SWAT does not allow modelling of riparian zones as wooded filter strips along streams

and does not allow application of two or more different filter strip values to one HRU at the same time. SWAT uses the same trapping efficiency for sediments and nutrients.

The filter strip trapping efficiency for sediment and nutrients in SWAT is calculated as (eq13):

$$trap_{ef} = 0.367 \cdot \left(width_{filtstrip}\right)^{0.2967} \quad \textbf{(eq13)}$$

Where: $trap_{ef}$ fraction of the surface flow constituent loading trapped by the filter strip,

$width_{filtstrip}$ width of the filter strip (m).

Filter strips can also reduce loads of contaminants in subsurface flow that passes through the strip. The trapping efficiency for subsurface flow components is calculated as (eq14):

$$trap_{ef,sub} = \frac{\left(2.1661 \cdot width_{filtstrip} - 5.1302\right)}{100} \quad \textbf{(eq14)}$$

Where: $trap_{ef, sub}$ fraction of the subsurface flow constituent loading trapped by the filter strip,

$width_{filtstrip}$ width of the filter strip (m).

The main positive sides of this scenario are reduction of sediments and nutrients in surface runoff by approximately 55% and that field buffer strips are already included in the active local stewardship subsidy payment scheme. This type of mitigation measure also does not require high cost educational courses but only a few presentations for the farmers. The main negative point is loss of cultivated land and consequently lower yields and incomes, as with installation of 4 m buffer strips approximately 16 % of a 1 ha productive field or pasture would be excluded from cultivation.

5.2 SCENARIO 2 – EXTENSIVE LAND USE MANAGEMENT

The extensive scenario (ELUM) was based on an interview with farmers who follow an organic based dairy farming regime. It is also based on the future socio-economic and land use scenarios defined by Morris (2003) as Local Stewardship or Local Community Agriculture with locally (national, regional) defined high support schemes reflecting local priorities for food production, income and environment (Table 2.5). It reflects local (on World scale) priorities and a focus on crops and products produced for the local market. The purpose of this scenario is not to simulate a full organic system, but rather a more extensive land use management, where farmers would replace an intensive crop such as maize with red clover or other legumes to reduce nitrogen application rates.

A new extensive rotation (red clover/red clover/winter wheat/winter barley) was set based on the interview at the organic farm (Appendix E9). Red clover is a short-lived perennial plant which will usually persist for two to four years under intensive management and can grow well in a wide range of soil conditions. Bacteria located in the root nodules convert atmospheric nitrogen into clover accessible nitrogen. This symbiosis provides high herbage yields from only modest inputs. Red clover has high nitrogen and consequently high protein content. This results in greater utilisation efficiency and reduced losses of nitrogen from the fields to surface waters and groundwater.

All crop rotations (ROT1, ROT2, ROT3 and ROT4), except the five years rye grass rotation (ROT5), were replaced with the new extensive rotation (ROTE – rc/rc/ww/wb) in four different starting crop sequences (Appendix F3). The five year rye grass rotation (ROT5) was changed to three years rye grass and two years red clover rotation (RT5E - rg/rg/rg/rc/rc) with three cut silage management (Appendix E10). Along with these new rotations the average stocking density for livestock was lowered. The baseline scenario livestock density was lowered from 4 LU ha^{-1} (PAS2 and PAS3) to 2 LU ha^{-1} for all dairy pasture land and sheep density was lowered from 2.25 LU ha^{-1} (PA11 and PA12) to 1.125 LU ha^{-1} (P11S and P12S) for all sheep pasture land. Note that animals are only

on half of the HRU-s at any one time, and are rotated between HRU-s. The manure deposited and biomass consumed (Table 5.3) was changed according to the recommendations proposed by Frame (1992). The fertilisation rates on sheep pastures were set to half of the baseline scenario. The three cut silage grass management (PSG4) nitrogen (N) fertiliser application rates were lowered from 270 kg ha^{-1} $year^{-1}$ to 170 kg ha^{-1} $year^{-1}$ as suggested in ABC (2006), and for that reason a new management file was created (PG4E). Other fertiliser application rates were calculated based on Fertiliser Recommendations (MAFF, 2000), ABC (2006), SAC (1999) for medium and low yields. All land management files and parameters (fertiliser application rates, dates, tillage operations) are presented in Appendixes E9 to E12.

Table 5.3: Grazing parameters for dairy cow and sheep per HRU, included in SWAT management files.

	BIO_EAT	MANURE_KG	BIO_MIN
Animal type	**kg DM ha^{-1} day^{-1}**	**kg DM ha^{-1} day^{-1}**	**kg DM ha^{-1} day^{-1}**
Dairy cow (4 LU ha^{-1} HRU^{-1})	66.00	15.00	200
Sheep (2.25 LU ha^{-1} HRU^{-1})	37.06	12.30	200

Key: DM – Dry Matter; LU – Livestock Unit

As in scenario 1, a value of 4 m was used for the width of edge-of-field filter strip parameter (FILTERW).

The positive and negative sides of field buffer strips are presented under scenario 1. The main positive side of this extensive scenario is reduction of sediments and nutrients in surface runoff and in river loads, due to the change in crop rotations, reduction of livestock grazing density and lower fertiliser application rates. On the positive side for the farmer this would result in lower costs for mineral fertilisers and consequently in lower production costs. The negative points are the costs of special subsidy payments for those farmers who would change arable rotations. Farmers would also have lower incomes as they would have to reduce dairy herds due to the exclusion of maize (high yields, protein values) consequently milk production would also drop. Changes in crop management would require educational courses for farmers and that would mean additional costs. The cost-benefit relationship could be balanced if farmers would enter

in to an organic farming scheme, where the subsidy payments and market prices of milk and meat are higher.

5.3 SCENARIO 3 – SHEEP LAND USE MANAGEMENT

The sheep land use management (SLUM) scenario is used to test how far land use and management might need to change to deliver water quality improvements to the river Axe catchment, whilst maintaining landscape in it's present state. This scenario is an extension of scenario 2 with lower mineral fertiliser application rates and manure deposited. It excludes all baseline crop rotations (ROT1, ROT2, ROT3 and ROT4), and replaces them with extensive five year rotations – ROTE (Appendix E9) as described in scenario 2. The five year rye grass rotation (ROT5) was changed to a two cut hay land management (PG1S) with low fertiliser application rates of 170 kg ha^{-1} $year^{-1}$ (ABC, 2006) and grazed aftermath with sheep grazing density of 2.25 LU ha^{-1} per HRU. Distribution of crop and grass rotations across the study area is presented in Appendix F4.

Along with the new rotation pattern the average density of livestock was changed. Baseline scenario dairy cow grazing (PAS2, PAS3) is excluded from the model and only sheep are considered to be present in the study area. A sheep stocking density of 1.125 LU ha^{-1} per total sheep pasture land use (Table 5.3), that is the same value as for the extensive scenario 2. With exclusion of dairy grazing, sheep grazing (P11S and P12S) is modelled also over land use which was previously assigned to dairy cow grazing (PAS2 and PAS3, respectively). Original baseline scenario fertilisation rates for sheep pastures were lowered by half. The baseline three cut grass silage land management (PSG4) is reduced to a two cut hay land management system (PG2S) with low fertiliser application rates of 170 kg ha^{-1} (ABC, 2006) and no grazing. Fertiliser application rates were calculated based on Fertiliser Recommendations (MAFF, 2000), ABC (2006), SAC (1999). All land management files and included parameters (fertiliser application rates, dates, tillage operations) are presented in appendixes E9 and E12 to

E14. As in scenario 1, a value of 4 m was used for the width of edge-of-field filter strip parameter (FILTERW).

The positive and negative sides of scenario 1 (field buffer strips) and 2 (extensive rotation, lower livestock density and fertilisation rates) measures are presented in previous sections. The positive side of this scenario is that even further reduction in sediment and nutrients in surface runoff and loads to surface water and groundwater can be expected, as livestock density is lowered and dairy farming which causes the major problems in the catchment, is excluded. Consequently this also requires less intensive pasture and hay field management, which means less mineral fertiliser costs. The negative sides are mainly connected to exclusion of cattle herds from catchment. This would mean that the majority of farmers would have to change the production and management and government agencies would need to support farmers with knowledge and subsidy payment schemes, to succeed in such drastic changes. All this would result in high cost. This scenario should be treated as a long transition process of changing management rather than a short time process, as it could provoke financial and social problems. As mentioned in scenario 2 farmers could be included in the organic subsidy payments schemes that could help them to overcome drastic changes in land and livestock management. Great care would be needed as the priority is that farmers would farm on an extensive level and not stop cultivating the land as this would also change the environmental and tourist value of the area, which is mainly, based on the landscape characteristics.

6 RESULTS AND DISCUSSION

To meet the aim of this research the baseline and defined land use management scenario simulations were run for a period of 20 years (1986-2005) including a 2 year (1986-1987) warm up period, which was excluded from the analyses. The SWAT output data has been compared between the baseline and three defined scenarios. To test the effect of diffuse source nutrients the baseline scenario was run without point sources. Comparison has been based on pollutant (total nitrogen, nitrate, total phosphorus, mineral phosphorus, sediment) inputs and outputs at the catchment and sub-catchment level, and at the outlets of the four catchment sections (River Yarty, River Coly, Upper Axe, and Lower Axe). The results are presented as sub-catchment contributions of pollutant yield in surface runoff and as river loads for each of the catchment sections. Explanation of the baseline scenario results includes presentation of study period average annual sub-catchment yields of surface runoff and loads of diffuse pollutants in the river with spatial and temporal variability. Explanation of the new scenario results include comparison of changes in the study period average annual loads of diffuse pollutants entering and leaving the catchment as percentages of the baseline scenario. Also average monthly load and concentration comparison at the main catchment outlet between scenarios has been performed. Spatial results of SWAT model runs are produced for baseline and land use scenarios to obtain visual representations of inputs and outputs of diffuse nutrients from each of the 25 sub-catchments. Finally, based on SWAT results the impact of defined scenarios on diffuse nutrients will be evaluated and possible mitigation measures recommended that could deliver water quality improvements.

6.1 BASELINE SCENARIO

6.1.1 Sub-catchment contribution

The baseline scenario shows high spatial and temporal variability in sediment and nutrient yields in surface runoff for the 25 sub-catchments (Table 6.1, 6.2, 6.3). Temporal variability for the baseline scenario at the main catchment outlet indicates that there is on average a 43% chance of an annual sediment yield of 5.08±2.53 t ha^{-1} $year^{-1}$, a 40% chance of a total nitrogen yield of 13.85±4.39 kg ha^{-1} $year^{-1}$ and a 46% of a total phosphorus yield of 3.10±1.17 kg ha^{-1} $year^{-1}$. Different weather conditions and land management practices from year to year are the main cause of temporal variability. On the other hand differences in soils, slope, and land use between sub-catchments are the main cause for spatial variability which is higher than temporal. The spatial variability is calculated as the standard deviation of the average annual values obtained for the each of the 25 sub-catchments separately. Spatial variability indicates that there is on average a 30% chance of an annual sediment yield of 5.08±2.81 t ha^{-1} $year^{-1}$, a 33% chance of a total nitrogen yield of 13.85±5.60 kg ha^{-1} $year^{-1}$ and a 32% chance of an total phosphorus yield of 3.10±1.46 kg ha^{-1} $year^{-1}$. High spatial variability is reasonable as spatial variability of topography (slope), soils, land use and land management is high across the study area.

Table 6.1: Baseline average annual contribution of sediment yield in surface runoff transported into the reach for the simulation period and variability among catchment section sub-catchments.

Catchment section	Average Sediment Yield (t ha^{-1} $year^{-1}$)	Spatial Variability (Stdev) (t ha^{-1} $year^{-1}$)	Temporal Variability (Stdev) (t ha^{-1} $year^{-1}$)
River Yarty	0.91	0.48	0.43
River Coly	4.23	2.69	2.59
Upper Axe	5.42	2.77	2.44
Lower Axe	8.01	4.54	4.36
Total Catchment	**5.08**	**2.81**	**2.53**

Table 6.2: Baseline scenario average annual total nitrogen yield in surface runoff transported into the reach for the simulation period and variability among catchment section sub-catchments.

Catchment section	Average Total Nitrogen Yield (kg ha^{-1} year^{-1})	Spatial Variability (Stdev) (kg ha^{-1} year^{-1})	Temporal Variability (Stdev) (kg ha^{-1} year^{-1})
River Yarty	5.56	2.78	2.16
River Coly	13.36	5.68	5.08
Upper Axe	13.60	5.50	4.43
Lower Axe	20.19	7.60	6.45
Total Catchment	**13.85**	**5.60**	**4.39**

Table 6.3: Baseline average annual contribution of total phosphorus yield in surface runoff transported into the reach for the simulation period and variability among catchment section sub-catchments.

Catchment section	Average Total Phosphorus Yield (kg ha^{-1} year^{-1})	Spatial Variability (Stdev) (kg ha^{-1} year^{-1})	Temporal Variability (Stdev) (kg ha^{-1} year^{-1})
River Yarty	1.07	0.55	0.44
River Coly	2.71	1.31	1.14
Upper Axe	3.10	1.46	1.17
Lower Axe	4.76	2.18	1.86
Total Catchment	**3.10**	**1.46**	**1.17**

6.1.2 River loads

The sediment and nutrient transported in the river were calculated as average yearly loads for the simulation period. Loads are a combination of a range of factors such as surface flow, subsurface flow, groundwater flow, point sources of discharge and river capacity (size, slope). The outlets of four catchment sections (River Yarty, River Coly, Upper Axe and Lower Axe) are located at the SWAT sub-catchments 21, 12, 4 and 25, respectively (Table 6.4). All sediment and nutrient loads transported by the tributaries drain through the channel outlet and as such represent contributions to the outlet. The loads at outlet 25 are total loads transported from the entire catchment. Results for the baseline scenario show that average sediment load carried out of the catchment is 9,167 t year^{-1}. At the same time results show that not all sediment that enters the river channel

is carried out of the catchment. The reason is in deposition of sediment on flood plains and in the meandering river channel in the lower Axe sections as a consequence of low river flow velocities. When flow velocity in the river channel is low sediment will be deposited and not transported downstream. Total phosphorus loads in the river do not show this kind of behaviour. The SWAT optional feature called in-stream nutrient processes (QUAL2E), which includes also organic phosphorus settling, was not used due to the low agreement between measured and simulated phosphorus concentrations for calibration and validation of the model. Migliaccio et al. (2007) in their research, evaluating landscape and in-stream modelling to predict watershed nutrient yields, states that many of the SWAT model parameters affecting in-stream transport of nitrate and phosphorus are considered static over time and that the model should be refined to enable a user to make these parameters dynamic to accurately reflect seasonal variability.

Table 6.4: The baseline scenario average annual sediment load, total nutrient loads and river flow at the outlets of four catchment sections for the simulation period.

Catchment section (Sub-catchment)	Sediment (t year^{-1})	Total Nitrogen (kg year^{-1})	Total Phosphorus (kg year^{-1})	River Flow (m^3 s^{-1})
River Yarty (21)	4,247	195,811	8,195	1.62
River Coly (12)	7,122	436,612	23,385	1.63
Upper Axe (4)	11,635	690,873	61,859	3.18
Lower Axe (25)	9,168	1,670,057	121,518	7.37

6.2 LAND USE MANAGEMENT SCENARIOS

6.2.1 River Flow

Changes between baseline and land use management scenarios average annual base flow (groundwater flow) and surface runoff are small, which is reflected in the total flow (Table 6.5). Both FBS and ELUM scenarios reduce annual average water yield, groundwater contribution and surface runoff, and increase annual average evapotranspiration. ELUM scenario always has a larger impact than FBS scenario. In

contrast, SLUM scenario increases all flow components and reduces evapotranspiration (ET).

Table 6.5: Impacts of the scenarios on average annual water yield and contributions to stream flow and actual evapotranspiration from the average sub-catchment in *.sub output file for the simulation period.

Scenario	Precipitation (mm)	Total Water Yield (mm)	Groundwater (mm)	Surface runoff (mm)	ET (mm)
BASE	1063.08	605.81	331.25	201.57	384.08
FBS	1063.08	605.15	330.77	201.44	384.84
ELUM	1063.08	602.65	329.96	199.23	387.80
SLUM	1063.08	616.96	338.54	203.87	371.86

Average annual daily river flow for the simulation period at the outlet of the four catchment sections was affected very little for the three scenarios. The greatest change in average annual daily river flow was observed in the SLUM scenario (Table 6.6) and especially at the River Yarty section outlet where river flow increased by +5.88%. Other two scenarios (FBS, ELUM) results show only a small change in river flows. A further scenario with point source discharge exclusion was carried out to check their influence on river flow and quality. Results show a reduction of -4.20% in daily river flow at the Axe outlet compared to the baseline scenario. Point sources have the biggest effect on river flow in the River Coly section (-8.63%) and the lowest effect on the River Yarty section (-0.15%). River flow in the Upper Axe is reduced by -2.15% with exclusion of point sources. Small changes can be observed from Figure 6.1 which shows changes in average daily river flows for management scenarios as a percentage of the baseline scenario.

Table 6.6: Impacts of the scenarios at the four catchment section outlets, percentage of change in average annual river flow from baseline scenario.

Catchment Section Outlet	Scenario - River Flow Change (%)			
	FBS	**ELUM**	**SLUM**	**No Point Sources**
River Yarty	0.00	+1.45	+5.88	-0.15
River Coly	-0.01	-0.28	+2.17	-8.63
Upper Axe	-0.18	-0.59	+1.88	-2.15
Lower Axe	-0.10	-0.32	+2.53	-4.20

These small changes indicate that land use management change and introducing mitigation methods do not affect river flow at a noticeable level. This was expected as selection of the mitigation methods was targeted to sediment and nutrient reduction. Small differences in the FBS scenario were expected as field buffer strips in general reduce only pollutants and not water flows. Small differences in the ELUM scenario can be related to the change in crop rotations as red clover develops deeper roots than rye grass. Differences in the SLUM scenario are mostly related with the transition from mixed dairy/sheep land management to the sheep only land management. This type of management requires less fertiliser and extensive grass management. This can affect grass turf and lead to a lower density of grassland. Extensive grass management leaves more organic matter on the surface which contributes to better soil conditions and causes less evapotranspiration and more percolation. This was especially observed in the River Yarty section where grassland predominates, and the highest changes for the river flow (+5.88% for the SLUM scenario) were simulated. However, higher surface runoff (SLUM scenario) could result in higher yields of nutrients transported by the surface flow into the river. Similar small effects on the river flow when simulating the BMP scenarios with SWAT were observed in Chaplot et al. (2003) in the Walnut Creek catchment where increasing (+40%) or decreasing (-60%) fertilisation rates did not have significant effects. When they changed all arable area to winter wheat production the river flow was increased by 8.3%. In contrast, with pasture throughout the catchment, mean annual river flow (0.48 $m^3 s^{-1}$) was lowered to 0.20 $m^3 s^{-1}$ (fertilised) and to 0.37 $m^3 s^{-1}$ (non fertilised). Bracmort et al. (2006) report that river flow was not affected by implementation of BMP-s in the Dreisbach and Smith Fry catchments and conclude that their impact on river flow at catchment scale is negligible.

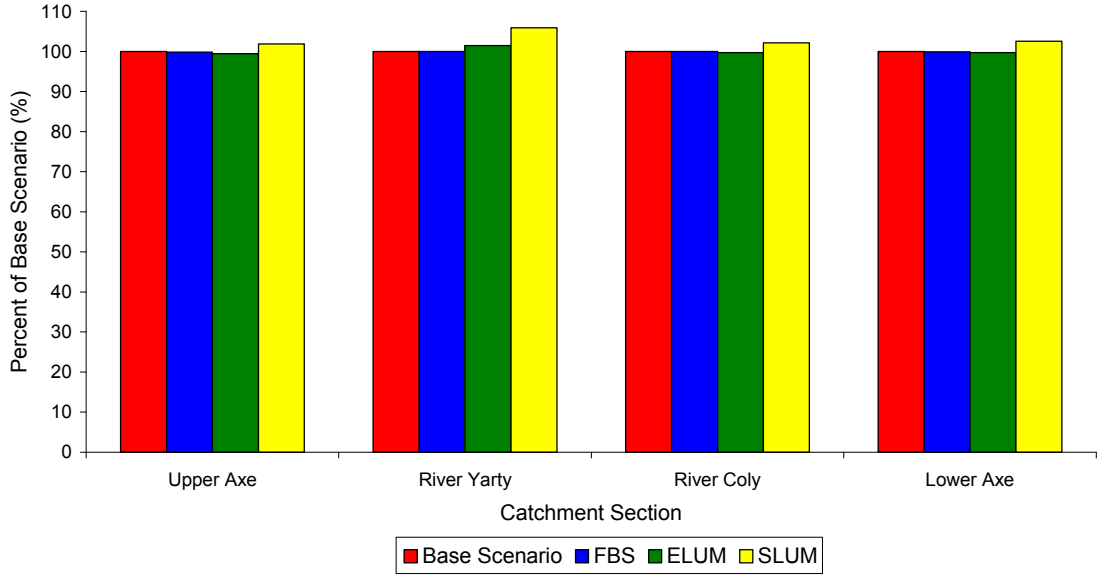


Figure 6.1: Average annual daily river flow change for the scenarios at the outlet of the four catchment sections as percent (%) of baseline scenario for the simulation period.

Figure 6.2 shows the average monthly daily river flow comparison between baseline and management scenarios for the simulation period at the main catchment outlet. This comparison confirms the previous results which show very small differences between management scenarios.

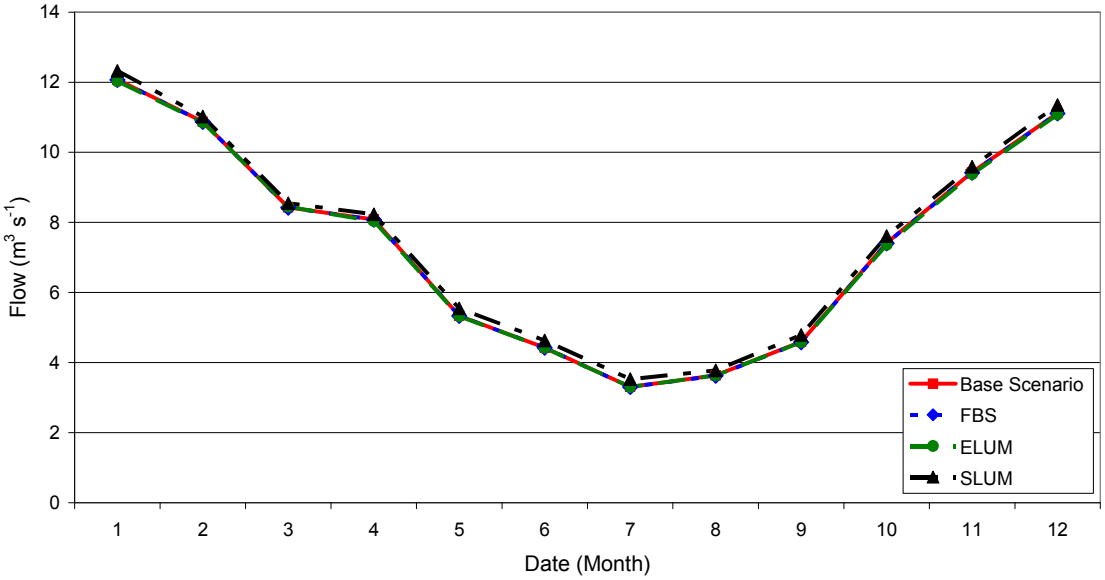


Figure 6.2: Comparison of average monthly daily river flow ($m^3 s^{-1}$) between baseline and management scenarios for the simulation period at the river Axe main catchment outlet.

6.2.2 Sediment

The reduction in sub-catchment contribution of sediment yield from all three scenarios was high (Table 6.7). The average annual reduction of sediment yield from the sub-catchments that is transported by surface runoff into the reach during the simulation period from FBS, ELUM and SLUM scenarios was -53.35%, -75.59% and -77.17%, respectively for the total catchment. This was in line with expectations. The highest reduction for the FBS scenario was simulated in the Lower Axe section, followed by the River Yarty, the Upper Axe and the River Coly. The ELUM scenario results show the highest reduction in surface flow sediments for the Lower Axe, followed by the Upper Axe, the River Yarty and the River Coly. For the SLUM scenario the highest reduction was again simulated in the Lower Axe, followed by the River Yarty, the Upper Axe and the River Coly. The highest reduction in surface runoff sediments for all three scenarios was simulated in the Lower Axe section. This is mainly the consequence of the high percent of arable land simulated in this section, so, the introduction of mitigation measures had a significant effect on pollutant content in surface runoff.

Table 6.7: Impacts of the scenarios for the four catchment sections, as percentage change in average annual sediment yields in runoff from the baseline scenario.

Catchment section	Scenario – Sediment Yield Change (%)		
	FBS	ELUM	SLUM
River Yarty	-53.85	-73.63	-79.12
River Coly	-51.06	-68.79	-70.21
Upper Axe	-53.69	-76.20	-77.86
Lower Axe	-53.93	-78.15	-79.15
Total Catchment	**-53.35**	**-75.59**	**-77.17**

Figure 6.3 shows the spatial comparison of average annual sediment yield from the sub-catchments between baseline and management scenarios for the simulation period. From this figure and Table 6.7 it can be observed that introduction of mitigation measures significantly reduces sediment transported in surface runoff to the main channel.

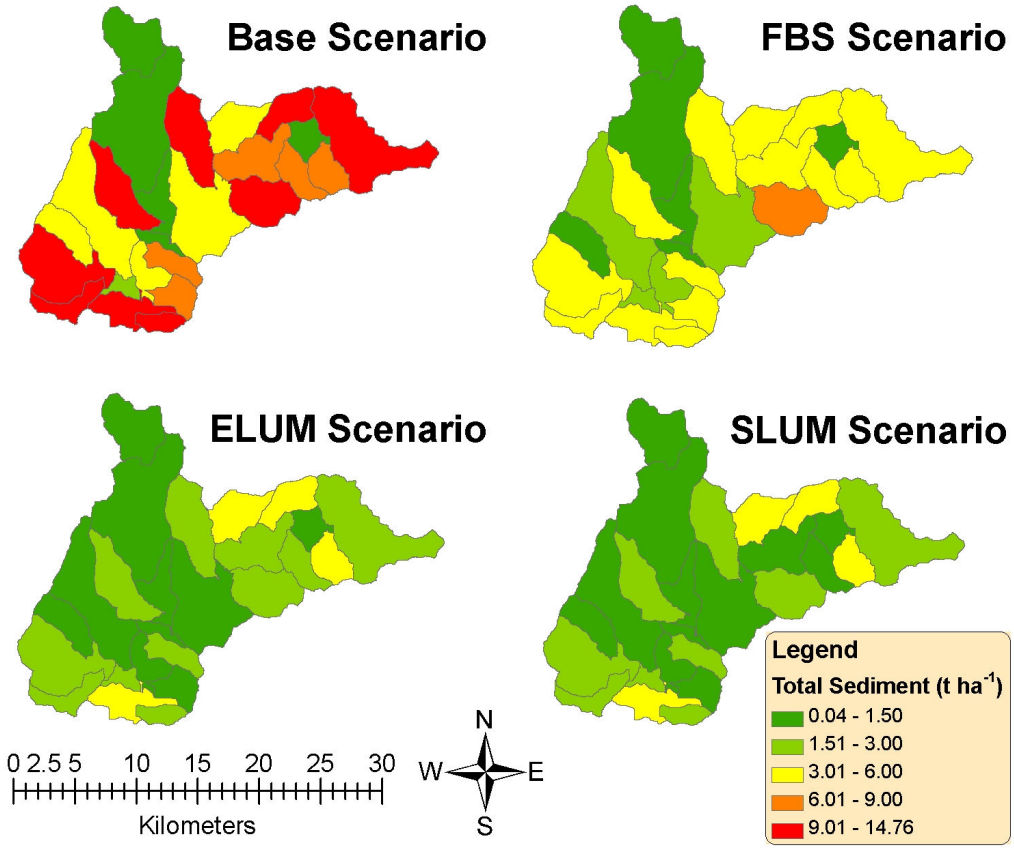


Figure 6.3: Comparison of average annual sediment yield (t ha^{-1} year^{-1}) transported from the HRU into the main channel for the simulation period, between baseline and land use management scenarios.

In contrast, the high percentage reduction of the sediment transported to the reach did not result in reduction of sediment load and concentration at the main catchment outlet. The reduction of average annual sediment loads transported with water out of the catchment during the simulation period for FBS, ELUM and SLUM scenarios was -3.49%, -7.34% and -5.58% respectively (Table 6.8). There were big differences between the four sub-catchment sections considered. From Figure 6.4 changes in average annual load of sediment in river for management scenarios as a percentage of the baseline scenario can be observed. To test the impact of point sources on sediment yield in the catchment an additional scenario without point sources was run. The reduction in sediment yield for the four catchment sections as a result of removing point sources was -0.05% for the River Yarty, -3.01% for the River Coly, -1.37% for the Upper Axe, and at main catchment outlet (Lower Axe) -3.50%. The simulation results

show small impact of point sources on average annual river loads of sediment in the river Axe.

Table 6.8: Impacts of the scenarios at the four catchment section outlets, percentage change in average annual river loads of sediment from the baseline scenario.

Catchment section Outlet	Scenario – Sediment Load Change (%)			
	FBS	ELUM	SLUM	No Point Sources
River Yarty	-42.61	-52.21	-64.54	-0.05
River Coly	-14.61	-24.57	-27.38	-3.01
Upper Axe	-4.17	-14.81	-14.95	-1.37
Lower Axe	-3.49	-7.34	-5.58	-3.50

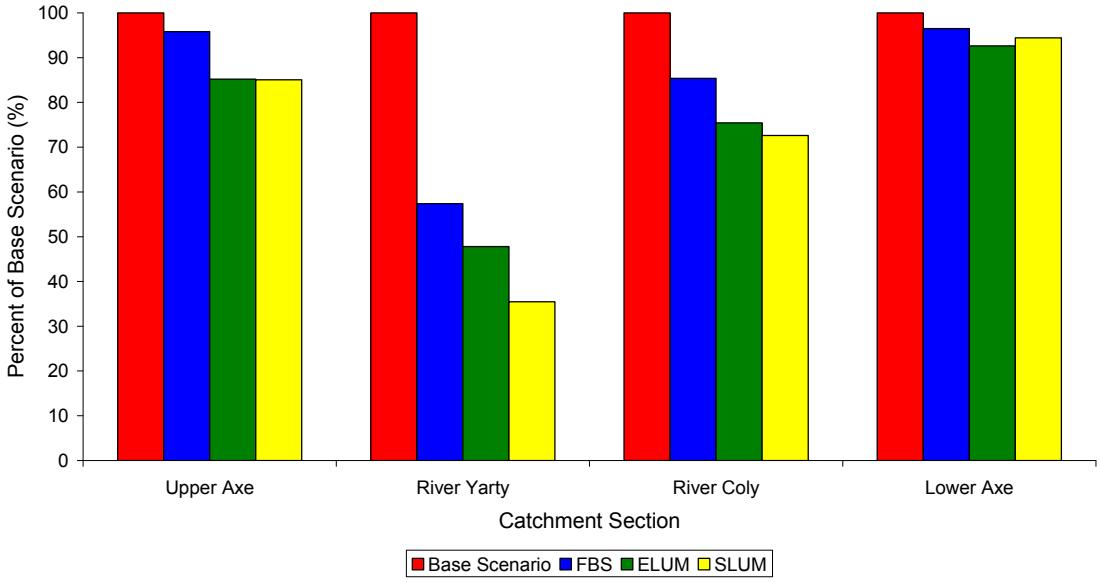


Figure 6.4 Change in average annual load of sediment at the outlet of the four catchment sections as percent of baseline scenario for the simulation period.

A plausible explanation of low overall reduction in sediment load transported with water out of the catchment lays in the deposition of sediment in the meandering lower section of the river, which happens in both baseline and management scenarios. All sediment which is in exceedance of river capacity will be deposited and not transported downstream, when flow velocity in the river channel is lower than the setting velocity. Thus reductions in sediment delivered to this final model reach do not translate into reductions at the catchment outlet. This suggests that reduction in sediment load at the catchment outlet may not be an ecologically sensible target to work towards. It is more

sensible to look at each tributary separately. The highest reduction in sediment transported out of the reach was simulated for the River Yarty catchment, this is mainly due to the fact that this is a section with very step slopes and introduction of BMP-s (field buffer strips, lower livestock density and extensive grass management) can show much better results that in other flatter sections (Upper Axe, Lower Axe).

A visual comparison of simulated average monthly sediment concentration at Whitford gauging station (Figure 6.5) for the study period confirms the numerical results explained above. Results for Whitford are presented as this is the only station where a validated comparison can be made as the mean annual value of 57.4 mg l^{-1} for year 2002 was only reported for this station (Environment Agency, 2004-b). It can be clearly seen that the introduction of mitigation measures reduce sediment yields in surface runoff which reflects on sediment concentrations in the river Axe. As can be seen, the average monthly sediment concentration for the study period, for the greater part of the year, is still above the EA EQS target value (25 mg l^{-1}), although this is an improvement on the baseline scenario where sediment concentrations are in excess for the whole year. This can cause problems to the population of fish as their eggs, or spawning grounds, get covered by sediment (Environment Agnecy, 2004-b; Palmer, 2004). The major problem is limited to the autumn, winter and spring months (September – April) when average monthly concentrations exceed EA target value. With introduction of mitigation measures the period of time when concentration exceeds target values is shortened to eight (ELUM) and to seven (SLUM) months. Palmer (2004) states that wooded buffer strips in the Kit Brook sub-catchment reduce rill and gully erosion. It is detailed that overland flow was traced as far as the wooded edge, where rills disappeared. Further, sediment and potential water pollutants might be filtered and absorbed within the woodland soils and provide better quality of river water for survival of fish stock. Due to the poor land use and soil map resolution and limitations of SWAT to simulate wooded buffer strips in certain areas of the catchment, it was not possible to simulate this mitigation measure.

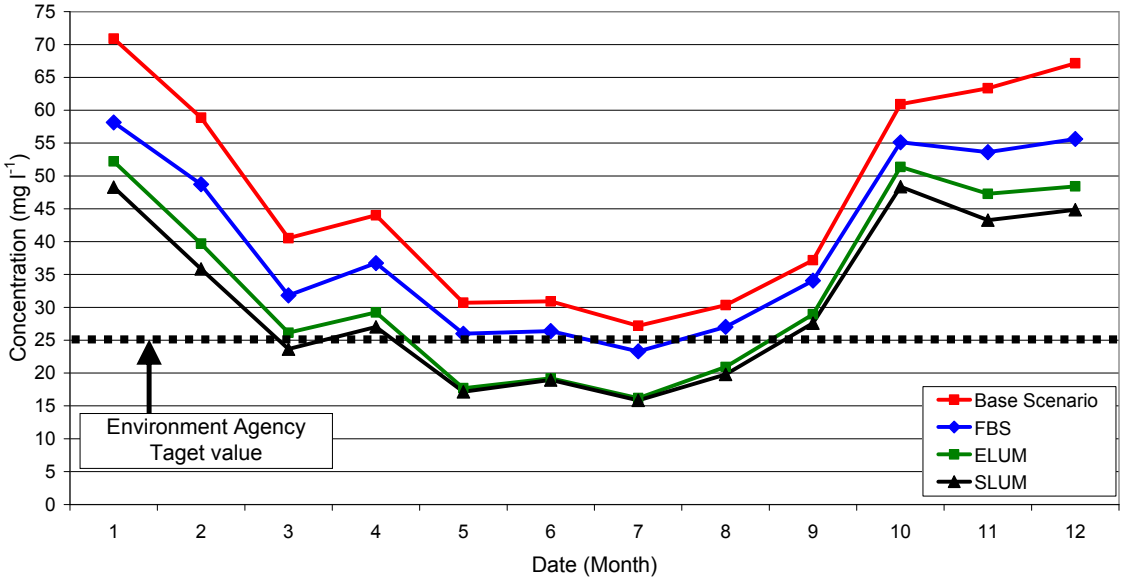


Figure 6.5: Comparison of average monthly sediment concentrations (mg l^{-1}) between baseline and management scenarios for the simulation period at the Whitford gauging station – sub-catchment 10 outlet.

6.2.3 Nitrogen

The average reduction of total nitrogen (ORGN and NSURQ) transported by surface runoff from sub-catchments into the river during simulation period was high for all three scenarios (Table 6.9). Results show a high reduction in average annual total nitrogen transported with surface flow in to the river when buffer strips (FBS) were simulated and an additional reduction of 16.24% with changed crop rotations (red clover) and lower livestock densities. The high reduction of total nitrogen for the FBS scenario is mainly associated with organic nitrogen reduction in surface flow, as this represents on average 93% of all nitrogen transported by surface flow in the baseline scenario. Further reductions of total nitrogen in the ELUM and SLUM scenarios are a consequence of mitigation measures (changed stocking density, crop types in rotations, tillage operations, and fertilisation rates). The spatial distribution of baseline and land use management average annual inputs and outputs of nitrogen from the sub-catchments for the study period can be observed in Appendices M1 and M2.

Table 6.9: Impacts of the scenarios for the four catchment sections, as percentage change in average annual total nitrogen yields in runoff from the baseline scenario.

Catchment Section	Scenario – Total Nitrogen Yield Change (%)		
	FBS	ELUM	SLUM
River Yarty	-55.04	-67.63	-73.20
River Coly	-51.20	-63.47	-65.72
Upper Axe	-53.97	-72.50	-76.84
Lower Axe	-54.33	-70.88	-72.86
Total Catchment	**-53.65**	**-69.89**	**-73.07**

The reduction in total nitrogen (ORGN_OUT, NO3_OUT, NH4_OUT, NO2_OUT) river loads at the main catchment outlet (Lower Axe) are shown in Table 6.10 The baseline scenario model was also run without point sources of discharge and results are shown in Table 6.10. This indicates that total nitrogen and nitrate loads from point sources do not represent a major threat to the river quality.

Table 6.10: Impacts of the scenarios at the four catchment section outlets, percentage of change in average annual total nitrogen river loads from baseline scenario.

Catchment Section Outlet	Scenario – Total Nitrogen Load Change (%)			
	FBS	ELUM	SLUM	No Point Sources
River Yarty	-14.90	-13.73	-18.40	-0.86
River Coly	-17.70	-57.93	-54.88	-2.43
Upper Axe	-25.88	-24.13	-44.12	-6.68
Lower Axe	-21.21	-37.32	-45.02	-7.86

As mentioned above a high reduction of nitrogen in surface runoff does not mean a high reduction in river loads. This is a consequence of mineral nitrogen percolation through the soil profile to groundwater, and point sources of discharge. Groundwater contribution from the shallow aquifer to the river flow is an important factor as it represents 69% in the baseline scenario, so lower reductions in river loads were expected. The percent of average annual nitrogen groundwater loading from the HRU-s for the baseline scenario study period is high and represents 51% of all nitrogen transported in to the river. Another big contributor to nitrogen load in the river is organic nitrogen 34% in surface flow, followed by nitrogen in lateral flow (11%) and surface flow (2%).

From Figures 6.6 and 6.7 positive impacts of the mitigation measures used in the land use management scenarios can be seen. This is encouraging as nitrogen represents a moderate threat to the river Axe. Measured mean annual nitrate nitrogen (NO_3-N) concentration for the study period (1988-2005) was for Whitford 4.5 mg l^{-1} and for Axe Bridge 4.6 mg l^{-1}. These NO_3-N concentrations are moderately lower than the EA EQS target value for the river Axe (annual mean 6.8 mg l^{-1}).

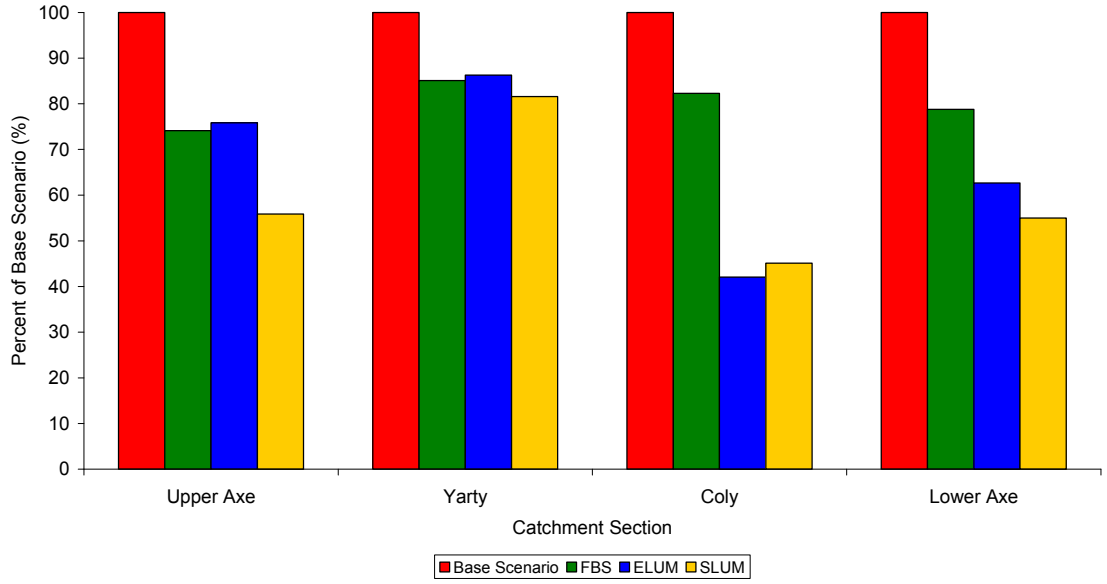


Figure 6.6: Change in average annual load of total nitrogen at the outlet of the four catchment sections as percent of baseline scenario for the simulation period.

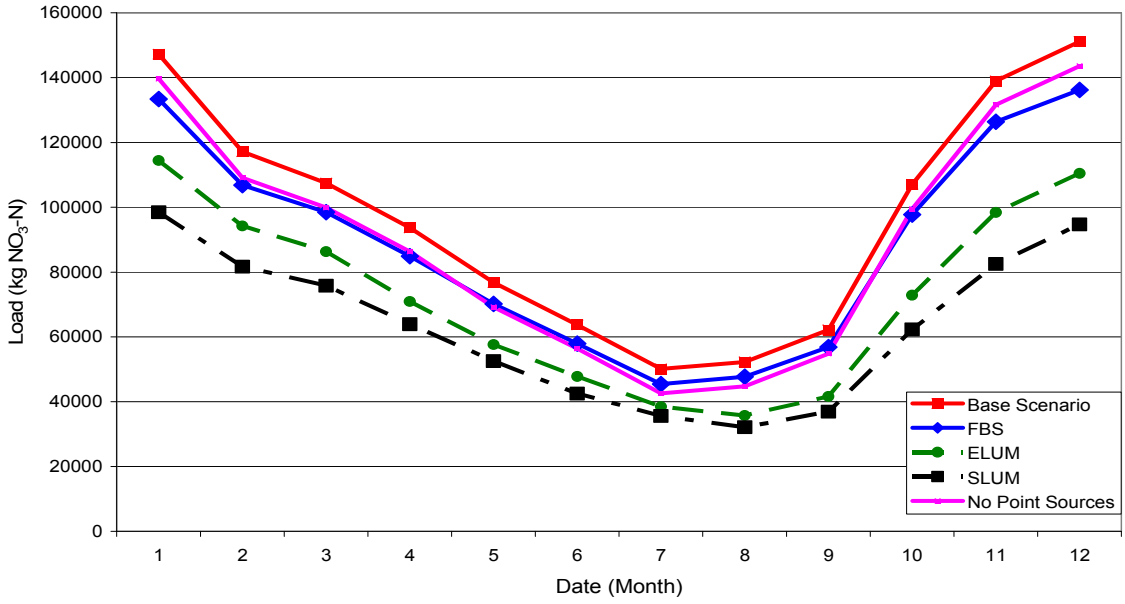


Figure 6.7: Comparison of average monthly nitrate nitrogen loads (kg NO_3-N $month^{-1}$) at the river Axe main catchment outlet between baseline and management scenarios for the simulation period.

6.2.4 Phosphorus

The reduction in total phosphorus (ORG_P, SOL_P, SED_P) transported by surface runoff in to the river during study period from sub-catchment is high for all three scenarios (Table 6.11).

Table 6.11: Impacts of the scenarios for the four catchment sections, as percentage change from the baseline scenario in average annual total phosphorus yield transported by surface runoff.

Catchment Section	Scenario – Phosphorus Yield Change (%)		
	FBS	ELUM	SLUM
River Yarty	-55.14	-61.68	-64.49
River Coly	-50.55	-61.25	-62.36
Upper Axe	-53.55	-70.00	-72.90
Lower Axe	-54.41	-72.27	-73.11
Total Catchment	**-53.55**	**-69.03**	**-70.65**

The high reduction of total phosphorus in the FBS scenario is mainly a consequence of reduction in organic phosphorus and to sediment attached phosphorus in surface flow. In the baseline scenario, organic phosphorus and sediment attached phosphorus represent on average 57% and 39% of the total phosphorus yield in surface runoff, respectively. Soluble phosphorus on the other hand represents only 5% of total phosphorus yield in surface runoff. A further reduction of total phosphorus in surface runoff is a consequence of the introduced mitigation methods (changed stocking density, crop types in rotations, tillage operations, and fertilisation rates). Results for high total phosphorus reduction in surface flow are in accord with sediment reduction in surface flow (Table 6.7). This confirms the importance of sediment attached phosphorus in total phosphorus yield in surface runoff and in river load. The spatial distribution of baseline and land use management average annual inputs and outputs of phosphorus from the sub-catchments for the study period can be observed in Appendix N1 and N2. They show the importance of the different mitigation measures used in this research. Although the phosphorus fertilisation rates stay almost constant through all three scenarios, mitigation measures succeed in reducing phosphorus yields in surface runoff significantly.

The reduction in total phosphorus river loads (ORGP_OUT, MINP_OUT) at the main catchment outlet (Lower Axe) are shown in Table 6.12. The baseline scenario model was also run without point sources of discharge. The results are also shown in Table 6.12.

Table 6.12: Impacts of the scenarios at the four catchment section outlets, as percentage change in average annual total phosphorus river loads from the baseline scenario.

Catchment Section Outlet	Scenario – Phosphorus Load Change (%)			
	FBS	ELUM	SLUM	No Point Sources
River Yarty	-52.49	-51.76	-56.35	-4.54
River Coly	-50.23	-61.47	-62.66	-6.26
Upper Axe	-48.98	-63.81	-66.05	-8.76
Lower Axe	-47.72	-60.58	-62.41	-11.12

As the majority of phosphorus in soils is attached to soil particles or organically bound with only a small portion soluble, reduction in total phosphorus loads in the river are much higher than in the case of nitrogen. Although these reductions are still less than those in surface runoff. High reductions under the FBS scenario show the importance of buffer strips at the edge of the field or pastures as a mitigation measure which can deliver important water quality improvements. Further reductions in phosphorus loads for the ELUM scenario are associated with changes in crops used in rotations which uptake phosphorus at a higher rate, and in reduction of livestock density. Reductions in the SLUM scenario show only a small improvement from the ELUM scenario and are associated with extensive grassland management with lower phosphorus fertilisation rates. Point sources of discharge (waste water treatment works) can not be the factor for the difference in reduction of phosphorus yield in surface runoff and loads in the river as they were simulated with the same rate of discharge for the baseline and three land use scenarios. Figure 6.8 shows changes in average annual total phosphorus loads at the four sub-catchment outlets for the management scenarios as a percentage of the baseline scenario. It shows high reductions in total phosphorus loads in the river.

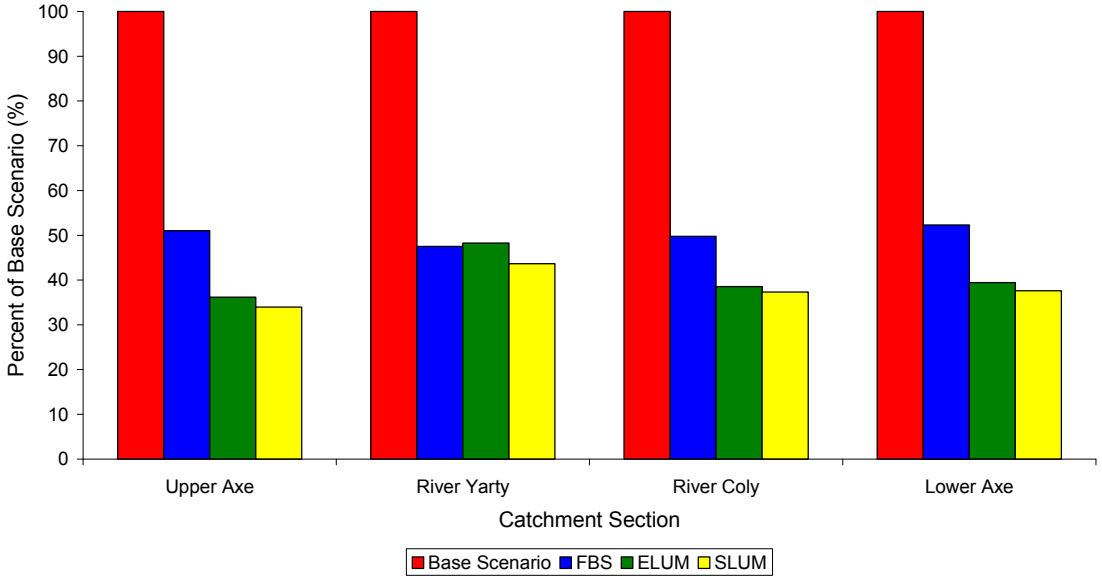


Figure 6.8: Change in load of total phosphorus at the outlet of the sub-catchments as percent of baseline scenario for the simulation period.

Figures 6.9 and 6.10 present a comparison between baseline and land use management scenarios average monthly mineral phosphorus load and concentration in the river at the main catchment outlet. It is interesting that average monthly mineral phosphorus loads for all three scenarios do not show the same reduction in average annual loads as for total phosphorus. Results for average phosphorus yield transported in surface runoff from sub-catchments into the river (Table 6.13) show that overall reduction in total phosphorus is a consequence of reduction in organic and sediment attached mineral phosphorus and not in a reduction of soluble phosphorus. Soluble phosphorus has in fact decreased in the FBS scenario, as organic and sediment attached did, but further changes in land use management cause only a reduction in organic and sediment attached, but not in soluble phosphorus yield which has in fact increased.

Table 6.13: Impacts of the land use management scenarios on average annual phosphorus yields transported in surface runoff into the river for the river Axe catchment.

	Average P Yield Transported in Surface Runoff (kg P ha^{-1} $year^{-1}$)			
Scenario	**ORGP**	**SOLP**	**SEDP**	**Total Phosphorus**
BASE	1.75	0.14	1.20	3.10
FBS	0.81	0.07	0.56	1.44
ELUM	0.48	0.11	0.38	0.96
SLUM	0.42	0.13	0.35	0.91

Table 6.14 show best results (highest reduction in mineral P) for the FBS scenario where only field buffer strips were simulated. It is interesting that further changes in land use management (ELUM, SLUM) do not bring any further reduction in mineral phosphorus loads or bring any improvements in river quality. This shows that reduction in total phosphorus may not relate to reduction of mineral phosphorus in the river. Therefore, care is needed as processes that reduce organic and sediment attached phosphorus can increase soluble phosphorus transported into the reach.

Table 6.14: Impacts of the land use management scenarios on average annual mineral phosphorus load and concentration in the river Axe at main catchment outlet and change as percent of base scenario load.

Scenario	Mineral P Load (kg year^{-1})	Change from Base Scenario (%)	Mineral P Concentration (mg l^{-1})
Base scenario	24561	0.00	0.11
FBS	17355	-29.34	0.08
ELUM	18558	-24.44	0.08
SLUM	19696	-19.81	0.08
No Point Sources	13681	-44.30	0.06

Phosphorus represents a high threat to the river Axe ecosystem as current levels exceed the standards developed under the Habitat Directives of 0.06 mg PO_4^{3-} l^{-1}. In 2006 annual mean ortho-phosphorus concentration at Whitford was 0.126 mg l^{-1} and at the Axe Bridge was 0.163 mg l^{-1} which is above the EA target. Simulation results show that mitigation measures contribute a significant reduction to the total phosphorus, but during most of the year phosphorus concentration still exceeds the target value of 0.06 mg l^{-1} of ortho-phosphorus (Figure 6.10). Table 6.14 also presents average annual mineral phosphorus load and concentration simulated without point sources. Results show that point sources represent 44.30% of all mineral phosphorus loads to the river, and that their exclusion would make a significant contribution to lowering concentrations of ortho-phosphorus in the river, as simulated mean annual mineral phosphorus concentration would reach the EQS target value of 0.06 mg l^{-1}. The importance of point sources can also be observed in Figures 6.9 and 6.10.

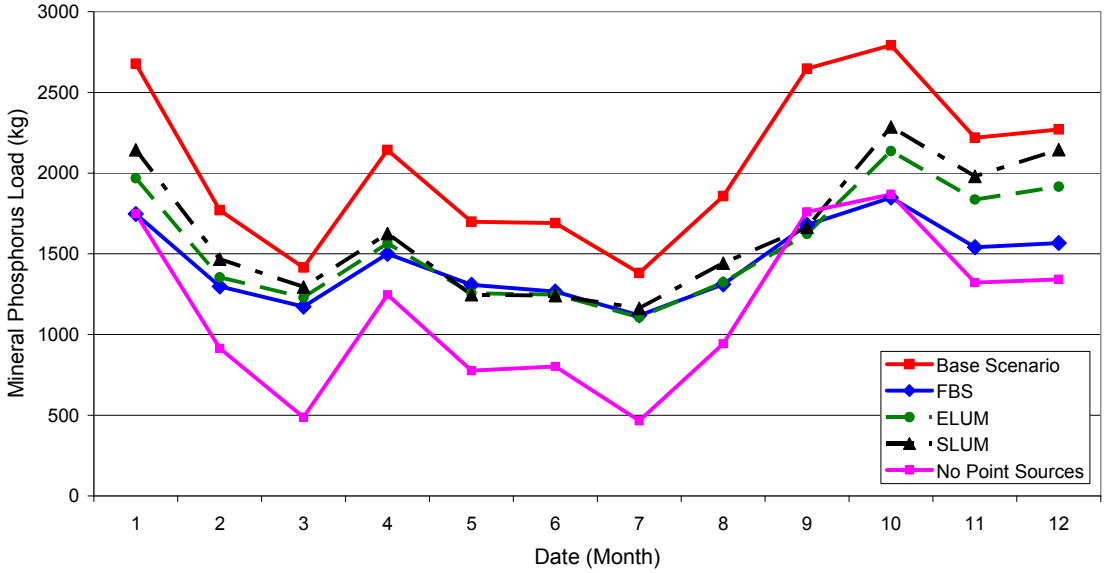


Figure 6.9: Comparison of average monthly mineral phosphorus loads (kg month^{-1}) in the river Axe at the main catchment outlet for baseline and land use management scenarios for the simulation period.

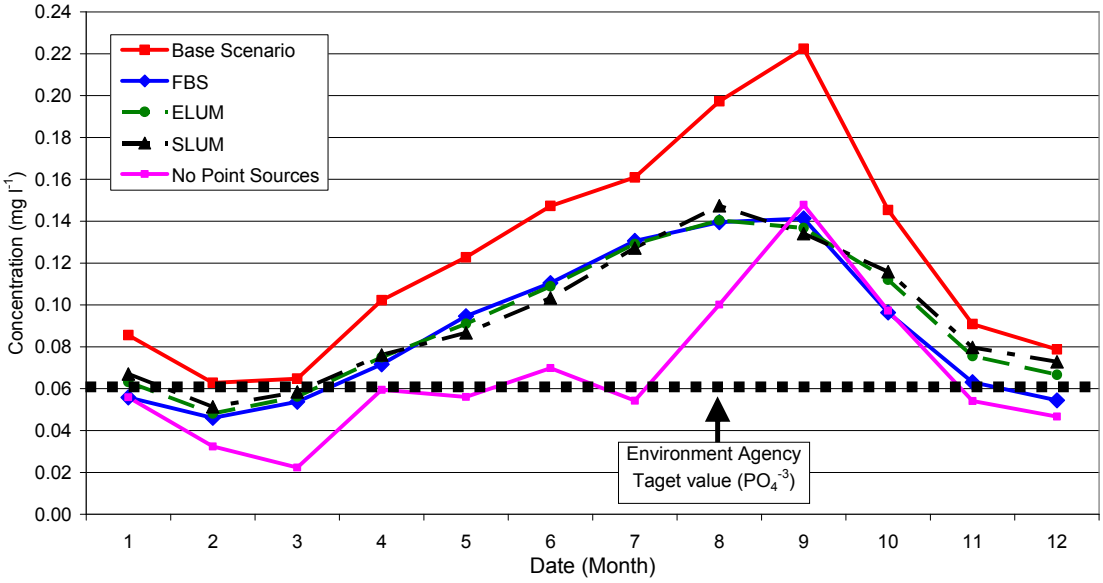


Figure 6.10: Comparison of average monthly mineral phosphorus concentrations (mg l^{-1}) at the river Axe main catchment outlet for baseline and management scenarios for the simulation period.

6.2.5 Scenario Evaluation and Recommendations

The evaluation of land use management is performed in the light of the European Union Water Framework Directive (WFD), Habitat Directive (HD) Special Areas of Conservation (SAC) and the English Catchment Sensitive Farming Delivery Initiative (ECSFDI). The goal of these Directives and initiatives is to deliver water quality improvements to the surface waters and groundwater, which should reach Good Ecological Status by 2015. As the river Axe is failing to reach WFD requirements the EA have set water quality targets known as Environmental Quality Standards (EQS). EQS give mean annual target values which should be reached to resolve river water quality problems. The EQS target value for nitrate nitrogen in the Axe is set at 6.8 mg NO_3-N l^{-1} or for nitrate 30 mg NO_3^- l^{-1}or grade 4, mainly due to the fact that nitrate does not represent a major threat to the river Axe and as the maximum nitrate levels for drinking water is 11.36 mg NO_3-N l^{-1} or 50 mg NO_3^- l^{-1} (MacDonald, 2005). The phosphorus EQS target value for the river Axe is set to 0.06 mg l^{-1} of total reactive phosphorus or grade 2 (Ash et al, 2005). The EQS for sediment is important for the river Axe catchment due to current high concentrations which cause problems for fish population and breeding; the current target value is set to 25 mg l^{-1} (Environment Agency, 2004-b). Scenario evaluation is made with reference to these EQS target values.

This evaluation was done with many model uncertainties in mind. The main uncertainties are in input data as spatial distribution of different crop rotations and land management practices, especially harvest and fertiliser application dates and rates which are difficult to set. The major reason for these uncertainties is in the soil map and land use map and hence their combination into HRU-s, which defines land management practices and their spatial distribution. It is also important to take into consideration that this is a catchment with a high dairy cow population (nitrogen) and with numerous STW with high discharge rates of phosphorus. Uncertainties can also originate in SWAT's direct linking of each HRU to the reach and not taking in to consideration the spatial position of the HRU in the sub-catchment.

Mitigation measures used for all three scenarios were found effective in reducing nutrient and sediment concentrations in the Axe catchment. They were found especially effective in reducing sediment and nutrients from surface flow; however reduction in average annual load in the river was lower than in surface runoff. This is mainly due to other diffuse (leaching to groundwater) or point sources (wastewater treatment works) of pollution and processes in the river. The reduction of nutrients and sediments in surface flow could also be overestimated as SWAT simulates reduction throughout the whole buffer strip, which is not the case in reality, as surface flow could concentrate in certain points of the buffer strip (Dillaha et al., 1989). Overall the reduction in nutrient yields in surface runoff and in the river is significant and it varies between sub-catchments as a consequence of different land uses, soil types, land management, stocking density, and topography combinations. It was also found that agriculture is the major contributor of nutrient pollution in rivers as point sources contribute only 7.86% total nitrogen and 11.12% of total phosphorus to the baseline scenario. However, considering only mineral phosphorus (ortho-phosphorus) in the river, point sources contribute 44.3% for the baseline scenario. The regulatory focus on point sources in the river Axe catchment is still high as further reduction could be achieved, especially from cheese factories and creameries. On the other hand reduction of pollutants from point sources will mean more emphasis on diffuse sources from agriculture. The mitigation measures contribute to noticeable total reduction of nutrients and sediment, but comparison of average monthly mineral phosphorus concentrations at the main catchment outlet for the simulation period shows that during most of the year phosphorus concentration still exceeds the EQS target value of 0.06 mg l^{-1} of ortho-phosphorus (PO_4^{-3}) (Figure 6.10). However, simulation shows that with total exclusion of point sources the EQS target value could be met.

Table 6.15 shows a summary of the impacts of the land use management scenarios for the main catchment outlet and for the Whitford gauging station in respect of Environmental Quality Standards for the river Axe. Results show that none of the scenarios would totally meet EQS and consequently the river would still fail to meet EU Water Framework Directive objectives. Results for the main catchment outlet show that sediments and NO_3-N already meet EQS for the base scenario and that introduction of

mitigation measures contributes to the further reduction of concentrations. However results for Whitford station show that only NO_3-N concentration meets EQS.

Although, the same mitigation measures contribute to the reduction of mineral phosphorus concentrations this is not enough to meet the EQS target value of 0.06 mg l^{-1}. Results for the Whitford station show that only NO_3-N meets EQS for all scenarios, due to the fact that concentrations are already moderate for the base scenario. Sediment concentration on the other hand shows a high reduction with introduction of mitigation measures in surface runoff yield and in sediment load in river, but still not enough to meet the EQS target value of 25 mg l^{-1} at Whitford gauging station (Figure 6.5). Mineral phosphorus concentration is reduced from 0.11 mg l^{-1} to 0.09 mg l^{-1} (FBS, ELUM) but for SLUM scenario a slight increase in concentration was simulated to 0.10 mg l^{-1}. Overall the river quality would improve as nitrate nitrogen and mineral phosphorus grade would improve from 4 to 3, and due to the considerable reduction in sediments. Results for simulation of the baseline scenario without point sources show significant improvement in mineral phosphorus concentration in the river as river quality would meet the EQS target value of 0.06 mg l^{-1} and would achieve quality grade 2.

Table 6.15: Comparison of baseline and land use management scenario impacts on mean annual sediment, mineral phosphorus and nitrate nitrogen concentrations (mg l^{-1}) for the study period compared with EQS target values.

	EQS target (Grade)	Scenario				
		Base	FBS	ELUM	SLUM	No Point Sources
Sub-catchment 10 outlet Whitford						
Sediment (mg l^{-1})	25.00	46.78	39.66	33.08	30.87	46.18
Mineral Phosphorus (mg l^{-1})	0.06 (2)	0.11 (4)	0.09 (3)	0.09 (3)	0.10 (3)	0.06 (2)
Nitrate Nitrogen (mg l^{-1})	6.8 (4)	4.5 (4)	4.1 (3)	4.3 (3)	3.4 (3)	4.1 (3)
Main catchment outlet						
Sediment (mg l^{-1})	25.00	22.32	20.67	19.02	18.69	21.55
Mineral Phosphorus (mg l^{-1})	0.06 (2)	0.11 (3)	0.08 (3)	0.08 (3)	0.08 (3)	0.06 (2)
Nitrate Nitrogen (mg l^{-1})	6.8 (4)	5.1 (4)	4.7 (4)	3.8 (3)	3.3 (3)	4.9 (4)

Key: (1-6) river quality grade

These results and evaluation of scenarios show that it will be difficult to reach the Environment Agency target value for ortho-phosphorus for the river as a whole. As scenarios were designed to maintain landscape as an important feature of the present

state, more extreme scenarios such as afforestation were not tested. This mitigation measure could reduce nutrient loads to the river, but would also change the water balance and could also reduce the present ecological value, quality of life and economic value of the catchment, especially in respect of tourism development. Results show that it is not possible to reach a large enough reduction in agricultural diffuse source nutrients to meet EQS target values and maintain landscape at the present state.

When recommending possible mitigation measures to deliver water quality improvements, careful evaluation and prioritisation of each measure has to be performed according to its positive and negative issues on the environment, agriculture, social life and economy.

Based on the simulated reduction of diffuse pollutants in the surface runoff and river loads, and economic effects of the scenarios on the farm incomes, the mitigation measures used in FBS and ELUM scenario were found to be the most viable. These two scenarios, especially the FBS, bring significant water quality improvements for the least cost, social consequences and time consumed for implementation. The SLUM scenario involving a change from dairy to sheep land use management did not result in enough water quality improvements to be considered viable, as cost and negative effects (social, financial) of the scenario are not justifiable.

At this point it is important to stress that the aim of this research was to define the impacts of land use management scenarios on diffuse source nutrients. This means that land use scenarios were not design to improve the problem with phosphorus originating from point sources, but only from diffuse sources. Therefore, two options are recommended to achieve significant improvements in the river Axe considering mineral phosphorus and sediment.

The first option is a combination of grass field buffer strips on arable fields and pastures and afforestation of agricultural areas along the river channel, which could work as riparian (forested) areas and effectively reduce diffuse pollution and stabilise river banks. Riparian areas are important because of soluble phosphorus and sediments

reduction in surface runoff. They could work as a sink for these two pollutants. In certain parts of the catchment (e.g. Kit Brook tributary) forested areas along the river channel are already recognised as an important measure in reducing surface erosion, as this tributary is one of few that can sustain fish population and enable fish breeding (Palmer, 2004). The positive points are a high reduction in diffuse pollution, subsidy payments for this mitigation measure already exist and there is no cost for special educational courses. The negative points are loss of cultivated land, which means lower yields and incomes, due to the fact that 4 m wide buffer strips mean approximately 16% less land in cultivation on a 1 ha field, although there is usually some existing buffer.

An alternative option is the combination of field buffer strips, lower livestock grazing density, reduction or total exclusion of maize and ryegrass from rotations and replacement with red clover. This would help to reduce sediment significantly, but for reduction of mineral phosphorus a further 50% reduction in phosphorus stripping from point sources should be introduced. A simple calculation shows that the combination of the FBS (low cost) scenario and reducing the contribution of point sources by half would bring significant water quality improvement to the river Axe catchment at the main outlet and meet the EQS target for ortho-phosphorus of 0.06 mg l^{-1}. The positive points of this combination of mitigation measures are a significant reduction in diffuse pollutants, lower cost of production and, with farmers inclusion in to the organic farming scheme, higher sale prices of milk and meat, and higher subsidy payments, the negative effect of lower incomes and special subsidy payments and costs for educational courses due to the change in management would be balanced. However, the cost for achievement of further phosphorus stripping from point sources of discharge, which can be substantial, must also be considered.

To deliver water quality improvements for the river Axe both types of pollution source (diffuse and point) will have to be addressed as results show that reduction only in diffuse does not meet the EA EQS target values which were set to meet the objectives of the EU WFD.

7 CONCLUSIONS

The Water Framework Directive (WFD) (Directive 2000/60/EC) requires new ecological standards for rivers, lakes and coastal waters by 2015 (European Parliament, 2000) and together with the Habitats Directive (HD) (Directive 92/43/EEC) and the Urban Wastewater Treatment Directive (UWTD) (Directive 91/271/EEC) sets new challenges for management of diffuse and point pollution sources. The diffuse sources are predominantly agricultural while point sources are a mix of sewage treatment works and industrial discharges. In the perspective of the WFD, in the UK DEFRA and the EA identified a number of catchments as priority sites for the English Catchment Sensitive Farming Delivery Initiative (ECSFDI) (DEFRA, 2002). This initiative aims to introduce a number of mitigation methods into farming practice to reduce contamination of rivers and water bodies.

The river Axe catchment is primarily agricultural, with agricultural land covering approximately 94% of the catchment, it is one of the ECSFDI priority sites, as intensive dairy farming and cultivation of high risk crops (maize) for fodder, cause problems with enhanced diffuse nutrients, especially phosphorus (Ash et al., 2005) levels in the river. There are however, a mixture of diffuse and point sources of pollution.

Water quality in 2006 was classified as grade 4 (moderate) for nitrate nitrogen (mean annual concentration 4.70 mg l^{-1}) and grade 4 (high) for ortho-phosphorus (mean annual concentration 0.16 mg l^{-1}) at the catchment outlet at Axe Bridge. The phosphorus concentration is above the EA EQS standard of 0.06 mg l^{-1} or grade 2 (Low). According to the EA (Ash et al., 2005) point sources contribute approximately 47 % of total ortho-phosphorus load. Another serious problem in the catchment is sediment which is closely related with the phosphorus problem, measurements in 2002 showed a mean annual suspended sediment concentration of 57.7 mg l^{-1} at Whitford, above the EA EQS standard of 25 mg l^{-1}.

For the purpose of this research the Soil and Water Assessment Tool (SWAT-2005) was used. Previous research shows that SWAT is becoming an important tool for assessing

catchment river water quality and agricultural diffuse pollution internationally. The model has also been found useful in the EU and UK context of addressing the objectives of the EU WFD, which requires identification of diffuse and point sources of pollution, their influence on the river ecosystem, and viable solutions for the studied catchment.

For the purpose of this research a wide range of spatial and temporal data was gathered starting with soil, land use and DEM maps, and associated parameters. Further, weather data (precipitation, temperature, wind, solar radiation, humidity) and river flow were obtained. For the representation of land use management livestock numbers, characteristic crops, rotations, fertilisation rates were gathered. To simulate appropriate levels of water and nutrients in the river, data for point sources of discharge and abstraction points were obtained for the study area.

The model has been run for a period of 20 years, from the 1st January 1986 until the 31st December 2005. The study period was divided into three periods of time: warm up (January 1986 until December 1987), calibration (January 1988 until December 1997) and validation (January 1998 until December 2005). Calibration was performed manually. Calibration and validation of the model, based on river flow, sediments, and nutrients (nitrate, ortho-phosphorus), was in the range for satisfactory model performance.

After calibrating the parameters the correlation between calibrated and simulated daily flow is relatively good giving a Nash-Sutcliffe efficiency coefficient (E_{NS}) of 0.62, correlation coefficient (R^2) of 0.62, root mean square error (RMSE) of 4.68 and PBIAS of 6.58. The performance statistics for the validation period are lower, R^2 of 0.53, RMSE of 7.24, PBIAS of 15.11; especially E_{NS} is low (0.47), but this is mainly due to the fact that flow variance during the validation period is significantly different from that during calibration. Statistical performance shows an overall underestimation of flow, especially of autumn and winter peak flows, and slow recession response in spring flows.

Sediment calibration showed good agreement between reported mean annual concentration of 57.7 mg l^{-1} and simulated mean annual concentration of 60.4mgl^{-1} for 2002. Sediment validation for the study area could not be performed due to shortage of data.

Daily calibration simulations showed high variability in the model outputs which is closely correlated with hydrological processes such as leaching to the shallow aquifer and surface runoff. More problems were encountered with nitrate nitrogen calibration and validation than with phosphorus. The problem was correlated mainly with the fact that the model was not able to simulate the seasonal nitrogen cycle accurately even when in-stream processes were used, although SWAT in-stream function models only algae processes and no macrophytes in the river and on the river banks, which are an important feature of the Axe catchment. Nitrate nitrogen is highly mobile and exact agreement is difficult to reach, even more because of the uncertainties and heterogeneity in spatial and temporal distribution of the crop rotations, stock density and fertiliser application rates. There are also large uncertainties in the soil and land cover parameters and their combinations. The high variability of daily outputs is also influenced by weather, especially rainfall events. All these reasons made concentrations of nitrate nitrogen in the model highly variable from day to day and year to year. An additional complication is that the Environment Agency takes water samples with mixed sampling frequency, on a monthly or fortnightly basis with some limited daily or hourly data, so is difficult to say if the model is over or under predicting nitrate nitrogen concentrations. The cumulative distribution of simulated and measured nitrate nitrogen in the river Axe showed that the simulated data includes more high and low values and does not return as many values in the mid-range as measured data. From this it could be concluded that measured data does not pick-up extreme values, because measured samples will have a tendency to under represent extreme flows.

Before 2002, when phosphorus stripping from discharges was introduced, the main source of phosphorus to the river was point discharges, therefore additional attention was given to point source parameters. As there were only small amounts of data available for estimation of nutrient concentrations in discharges, the results for the

calibration period were encouraging. Simulated phosphorus returns lower low values and higher high values, but not as many peak values as measured data. This is mainly due to the fact that some short-term sampling exercises were performed which resulted in monitoring of short-term high concentrations and because flow from STW discharges were simulated as constant daily loadings. That means that periods of time when discharge concentration or load of phosphorus in outflow were high or low are not modelled. The cumulative distribution results for the validation period for phosphorus show more high values than measured data, due to the reasons described for nitrate nitrogen calibration and validation.

The model also showed some shortcomings in modelling plant growth, as in certain years too much biomass was simulated and there were difficulties to simulate senescence. This could be the consequence of using default plant parameters, which are adopted for US conditions.

Three scenarios were developed to investigate possible different land use management impacts on the behaviour of diffuse source pollutants, with the purpose of delivering water quality improvements in the River Axe catchment. The purpose of scenarios was not to simulate drastic changes in the catchment to achieve reduction in nutrients, but rather to develop viable scenarios which would allow agriculture to operate at an appropriate level to maintain landscape in its present state. Scenarios and mitigation measures included were developed based firstly on interviews with farmers and the Environment Agency and on the basis of site investigation during a field trip to the study area, and secondly on the basis of different reports and action plans produced by DEFRA and the Environment Agency dealing with environmental problems in the River Axe catchment.

With the field buffer strips (FBS) scenario 4-metre field buffer strips for all HRU-s were simulated based on the DEFRA (2007-c) Entry Level Stewardship handbook. The extensive (ELUM) scenario was based on extensive land use management, where intensive crops such as maize and rye grass were replaced with red clover to reduce mineral nutrient application rates. At the same time average livestock density was

lowered and 4 m field buffer strips were installed. The sheep (SLUM) scenario included all mitigation measures used in the FBS and ELUM scenarios, with further reduction in mineral fertiliser rates and lower livestock density. In this scenario only sheep were considered to be present in the catchment.

SWAT's ability to represent mitigation methods and effectiveness of these measures in reducing nutrient concentration was based on long-term (18 years) mean annual and monthly values. This is due to the fact that short-term reduction may not be observed when comparison is based on year to year assessment, due to the variability in weather conditions. The study focused on the SWAT outputs from the sub-catchments and on the transport of pollutants at the outlets of the four catchment sections (River Yarty; River Coly, Upper Axe, Lower Axe). From the results it can be observed that SWAT is able to represent the proposed mitigation methods for all three management scenarios in the study area. Although some problems regarding nitrogen, especially the representation of seasonality of nitrogen concentration and unrealistic high variability in daily model outputs were observed, this did not impact strongly on the study. It was still possible to look at percentage change from the baseline scenario. Based on limited input data and numerous estimations of conditions and practices in the study area, SWAT was still found to be a useful and reliable tool for estimating impacts of management scenario mitigation measures to deliver water quality improvements to the river Axe.

Measures used in the study were found to be effective in reducing nutrient and sediment loads at the main outlet of the Axe catchment. Although, certain measures did not bring as much improvement as expected. Land use management scenarios reduced the average annual loads for total nitrogen at the outlet of the catchment by 21.21% (FBS), 37.32% (ELUM) and 45.02% (SLUM); 47.72% (FBS), 60.58% (ELUM) and 62.41% (SLUM) for total phosphorus, and 3.49% (FBS), 7.34% (ELUM) and 5.58% (SLUM) for annual average sediment yield. Based on the results it can be concluded that establishment of field buffer strips of 4m to all arable and grassland land use, exclusion of high risk crops (maize) and lowering the stock density on pastures would lower nutrient and sediment loads and bring significant water quality improvements to the

River Axe, although Environment Agency targets would still not be achieved for the river as a whole. However, the results also showed that significant improvements could be achieved with additional phosphorus stripping from point sources.

When recommending possible mitigation measures to deliver water quality improvements, careful evaluation and prioritisation of each measure has to be performed according to its positive and negative issues on the environment, agriculture, social life and economy. At this point it is important to stress that land use scenarios in this research were not design to improve the problem with phosphorus originating from point sources, but only from diffuse sources. Results show that none of the new land use management scenarios would solve the phosphorus problem. In this perspective only the FBS and ELUM scenario mitigation measures were found to be justifiable for implementation for the study area as they bring the most reduction in diffuse nutrients and sediments, for the least cost and social consequences. Therefore, two possible mitigation measure combinations, to deliver water quality improvements into the river Axe, are recommended. The first option is the combination of grass field buffer strips on field and pastures and afforestation of agricultural areas along the river channel, which could work as riparian (forested) areas and effectively reduce diffuse pollution and stabilise river banks. The second option is the combination of field buffer strips, lower livestock grazing density, reduction or total exclusion of maize and ryegrass from rotations and replacement with red clover. These would help to reduce sediment significantly, but for reduction of mineral phosphorus a further 50% reduction through phosphorus stripping should be introduced for point sources.

To deliver water quality improvements for the river Axe both sources (diffuse and point) of pollution will have to be addressed as results, based on model approximations, show that reduction only in diffuse source does not meet EA EQS target values which were set to meet the objectives of the EU WFD.

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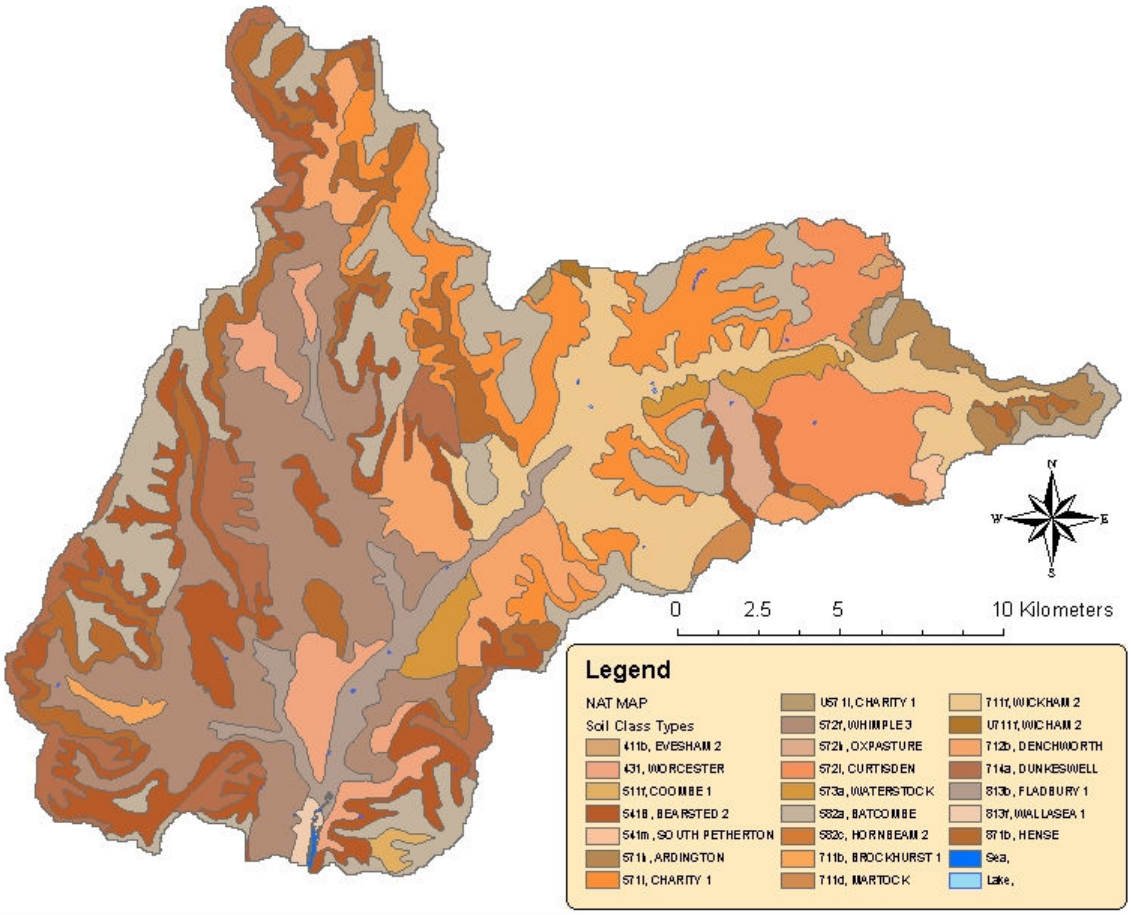
APPENDICES

APPENDIX A

Appendix A1: Soil map of the river Axe catchment (SSLRC, 1983).



APPENDIX B

Appendix B1: Precipitation data for nine gauging stations used in the SWAT model (BADC, 2007).

	Precipitation (mm)								
YEAR	Cricket Wildlife Park	Chard Junction	Wilmington	Farway Tedbridge	Pinhay	Seaton Junction	Wambrook	Forde Abbey	Buckland
1986	1091	1067	1217	1183	1013	1146	1178	1093	1178
1987	891	865	967	1034	897	916	934	852	882
1988	871	826	1051	1087	874	930	972	819	976
1989	995	953	1072	1022	875	948	1076	958	1042
1990	871	833	1043	957	792	881	945	814	882
1991	845	846	1008	992	916	929	1000	851	912
1992	863	810	1065	945	810	874	1041	810	924
1993	1138	1067	1217	1219	1111	1150	1141	1091	1073
1994	1150	1064	1264	1210	1167	1188	1253	1137	1121
1995	1208	1125	1208	1172	1038	1088	1217	1138	1116
1996	1048	943	1083	1019	932	949	1024	991	991
1997	1128	1066	1233	1239	1017	1136	1190	1085	1118
1998	1142	1078	1165	1187	1033	1126	1177	1102	1177
1999	1254	1196	1337	1260	1062	1242	1353	1188	1249
2000	1447	1267	1393	1350	1205	1260	1436	1310	1371
2001	1110	995	1181	1201	982	1045	1077	1023	1030
2002	1364	1189	1366	1301	1210	1237	1386	1364	1332
2003	844	761	918	896	841	876	865	782	760
2004	1122	973	1091	1066	959	964	1036	980	1035
2005	1024	1091	1050	1103	937	1017	1056	1050	1056
Average (1986-2005)	1070	1001	1146	1122	984	1045	1118	1022	1061
Average (1986-1997) (Calibration Period)	1065	1003	1162	1120	988	1056	1134	1021	1056
Average (1998-2005) (Validation Period)	1166	1067	1191	1168	1028	1092	1173	1100	1119

APPENDIX C

Appendix C1: Data obtained for SWAT model build.

	Data Required	Source
Weather data	Daily precipitation (rainfall)	BADC
	Temperature (max., min., wet and dry bulb for relative humidity)	BADC
	Wind speed	BADC
	Sunshine duration (for solar radiation)	BADC
Land management practices	Agriculture statistics	EDL
	Timing	DEFRA
	Amount applied (fertilisers, manure)	DEFRA
	Cropping regime	ADAS, DEFRA Best practice and standard crop rotation Interview with farmers
Abstraction	Ground water	EA
	Surface waters	EA
Discharges	Sewage and domestic discharges	EA
	Industrial discharge	EA
	Fish farms	EA
	DEM	CEH
	River network	Defined from DEM
	Land use	CORINE 2000
Soil	Spatial variability	NSRI
	Soil type	NSRI
	Properties	NSRI
Digital boundaries	Catchment	Defined from DEM
Calibration and Validation	Water quality	EA
	Flow data	EA

APPENDIX D

Appendix D1: General soil data in SWAT *.sol file.

SWAT Variable Symbol								
MUID	**SNAM**	**CMPPCT**	**NLAYERS**	**HYDGRP**	**SOL_ZMX**	**ANION_EXCL**	**SOL_CRK**	**TEXTURE**
WORCESTER	Soil-411b	0.00000	5	C	1500.00	0.500	0.500	C--ZC--C--ZC--C
EVESHAM 2	Soil-431	0.00000	4	D	1500.00	0.500	0.500	C--C--C--C
COOMBE 1	Soil-511f	0.00000	4	B	1500.00	0.500	0.500	ZCL--ZCL--CL--CL-
BEARSTED 2	Soil-541B	0.00000	5	A	1500.00	0.500	0.500	SL--SL--SL--SL--LS
SOUTH PETHERTON	Soil-541m	0.00000	5	A	1500.00	0.500	0.500	SL--SL--SL--SL--SL
ARDINGTON	Soil-571h	0.00000	5	B	1500.00	0.500	0.500	SCL--SL--ZCL--LS--LS
CHARITY 1	Soil-571l	0.00000	5	C	1500.00	0.500	0.500	ZCL--ZCL--ZCL--ZC--ZC
WHIMPLE 3	Soil-572f	0.00000	5	C	1500.00	0.500	0.500	CL--CL--C--ZCL-ZCL
OXPASTURE	Soil-572h	0.00000	5	C	1500.00	0.500	0.500	CL--CL--C--C--ZC
CURTISDEN	Soil-572i	0.00000	3	C	1500.00	0.500	0.500	ZL--ZL--ZL
WATERSTOCK	Soil-573a	0.00000	5	C	1500.00	0.500	0.500	ZCL--SCL--SCL--SCL--SL
BATCOMBE	Soil-582a	0.00000	4	C	1500.00	0.500	0.500	CL--ZCL--C--C
HORNBEAM 2	Soil-582c	0.00000	5	C	1500.00	0.500	0.500	CL--CL--CL--C--C
BROCKHURST 1	Soil-711b	0.00000	5	C	1500.00	0.500	0.500	CL--CL--C--ZC--ZC
MARTOCK	Soil-711d	0.00000	5	C	1500.00	0.500	0.500	ZCL--ZCL--ZC--ZCL--ZCL
WICKHAM 2	Soil-711f	0.00000	5	D	1500.00	0.500	0.500	CL--CL--C--C--ZC
DENCHWORTH	Soil-712b	0.00000	5	D	1500.00	0.500	0.500	C--C--C--C--C
DUNKESWELL	Soil-714a	0.00000	4	C	1500.00	0.500	0.500	ZCL--ZCL--C--C
FLADBURY 1	Soil-813b	0.00000	5	C	1500.00	0.500	0.500	C--C--C--C--C
WALLASEA 1	Soil-813f	0.00000	5	C	1500.00	0.500	0.500	C--C--C--ZC--ZC
HENSE	Soil-871b	0.00000	3	A	1500.00	0.500	0.500	OCL--OCL--SL
CHARITY 1U	Soil-U571l	0.00000	5	C	1500.00	0.500	0.500	ZCL--ZCL--ZCL--ZC--ZC
WICKHAM 2U	Soil-U711f	0.00000	5	D	1500.00	0.500	0.500	CL--CL--C--C--ZC

Appendix D2: Layer 1 soil data in SWAT *.sol file.

SWAT Variable Symbol												
MUID	SOL_Z1	SOL_BD1	SOL_AWC1	SOL_K1	SOL_CBN1	CLAY1	SILT1	SAND1	ROCK1	SOL_ALB1	USLE_K1	SOL_EC1
WORCESTER	200.00	1.08	0.21	45.30	3.60	38.00	43.00	19.00	2.00	0.09	0.14	0.00
EVESHAM 2	200.00	1.03	0.20	42.70	4.70	45.00	35.00	20.00	0.00	0.09	0.12	0.00
COOMBE 1	250.00	1.08	0.23	54.80	4.20	27.00	56.00	17.00	10.00	0.09	0.16	0.00
BEARSTED 2	200.00	1.17	0.22	113.50	2.90	13.00	31.00	56.00	10.00	0.09	0.13	0.00
SOUTH PETHERTON	200.00	1.18	0.23	116.30	2.60	9.00	38.00	53.00	0.00	0.09	0.14	0.00
ARDINGTON	200.00	1.12	0.21	116.30	4.00	21.00	15.00	64.00	5.00	0.16	0.11	0.00
CHARITY 1	150.00	1.37	0.19	16.50	1.40	26.00	58.00	16.00	15.00	0.09	0.19	0.00
WHIMPLE 3	250.00	1.10	0.21	67.20	3.60	28.00	42.00	30.00	10.00	0.09	0.13	0.00
OXPASTURE	250.00	1.06	0.22	77.30	4.70	29.00	34.00	37.00	10.00	0.16	0.13	0.00
CURTISDEN	250.00	1.35	0.21	29.90	1.60	15.00	66.00	19.00	10.00	0.23	0.19	0.00
WATERSTOCK	200.00	1.02	0.23	112.00	5.50	21.00	33.00	46.00	10.00	0.16	0.13	0.00
BATCOMBE	250.00	1.32	0.20	24.30	1.70	24.00	58.00	18.00	10.00	0.23	0.17	0.00
HORNBEAM 2	200.00	1.12	0.22	78.50	3.20	21.00	47.00	32.00	10.00	0.16	0.14	0.00
BROCKHURST 1	200.00	1.11	0.22	69.40	3.50	26.00	42.00	32.00	4.00	0.30	0.13	0.00
MARTOCK	250.00	0.94	0.25	73.90	4.50	30.00	63.00	7.00	5.00	0.09	0.24	0.00
WICKHAM 2	250.00	1.20	0.20	43.60	2.80	30.00	41.00	29.00	5.00	0.16	0.13	0.00
DENCHWORTH	150.00	0.97	0.21	51.80	5.60	43.00	40.00	17.00	10.00	0.16	0.13	0.00
DUNKESWELL	200.00	1.01	0.24	79.70	4.20	23.00	60.00	17.00	10.00	0.16	0.17	0.00
FLADBURY 1	150.00	0.93	0.21	45.30	7.10	50.00	30.00	20.00	0.00	0.09	0.12	0.00
WALLASEA 1	250.00	0.93	0.21	44.50	5.50	50.00	41.00	9.00	0.00	0.16	0.16	0.00
HENSE	200.00	0.77	0.27	93.30	15.00	29.00	36.00	35.00	10.00	0.09	0.13	0.00
CHARITY 1U	150.00	1.37	0.19	16.50	1.40	26.00	58.00	16.00	15.00	0.09	0.19	0.00
WICKHAM 2U	250.00	1.20	0.20	43.60	2.80	30.00	41.00	29.00	5.00	0.16	0.13	0.00

Appendix D3: Layer 2 soil data in SWAT *.sol file.

SWAT Variable Symbol												
MUID	SOL_Z2	SOL_BD2	SOL_AWC2	SOL_K2	SOL_CBN2	CLAY2	SILT2	SAND2	ROCK2	SOL_ALB2	USLE_K2	SOL_EC2
WORCESTER	300.00	1.30	0.17	14.40	1.90	37.00	50.00	13.00	0.00	0.00	0.17	0.00
EVESHAM 2	500.00	1.26	0.14	2.50	1.50	60.00	28.00	12.00	0.00	0.00	0.13	0.00
COOMBE 1	500.00	1.13	0.21	66.10	1.60	25.00	58.00	17.00	20.00	0.00	0.18	0.00
BEARSTED 2	450.00	1.27	0.19	159.80	1.10	8.00	35.00	57.00	10.00	0.00	0.17	0.00
SOUTH PETHERTON	500.00	1.28	0.23	116.30	1.00	8.00	39.00	53.00	0.00	0.00	0.17	0.00
ARDINGTON	300.00	1.08	0.02	235.90	2.80	15.00	14.00	71.00	5.00	0.00	0.11	0.00
CHARITY 1	450.00	1.42	0.17	14.40	0.50	27.00	60.00	13.00	15.00	0.00	0.25	0.00
WHIMPLE 3	400.00	1.43	0.16	35.20	1.20	24.00	42.00	34.00	10.00	0.00	0.15	0.00
OXPASTURE	450.00	1.45	0.15	20.00	0.70	30.00	43.00	27.00	10.00	0.00	0.17	0.00
CURTISDEN	500.00	1.43	0.19	29.90	0.50	15.00	68.00	17.00	10.00	0.00	0.26	0.00
WATERSTOCK	350.00	1.23	0.18	120.60	1.80	22.00	24.00	54.00	10.00	0.00	0.13	0.00
BATCOMBE	550.00	1.35	0.19	27.80	0.60	23.00	64.00	13.00	10.00	0.00	0.26	0.00
HORNBEAM 2	350.00	1.42	0.17	41.80	0.80	20.00	49.00	31.00	10.00	0.00	0.18	0.00
BROCKHURST 1	450.00	1.45	0.16	27.00	0.90	26.00	44.00	30.00	10.00	0.00	0.17	0.00
MARTOCK	450.00	1.37	0.18	13.40	1.50	30.00	58.00	12.00	5.00	0.00	0.20	0.00
WICKHAM 2	400.00	1.45	0.15	29.90	0.90	26.00	42.00	32.00	10.00	0.00	0.16	0.00
DENCHWORTH	500.00	1.21	0.14	0.80	1.90	64.00	30.00	6.00	0.00	0.00	0.16	0.00
DUNKESWELL	500.00	1.26	0.20	35.20	1.10	25.00	63.00	12.00	10.00	0.00	0.24	0.00
FLADBURY 1	500.00	1.20	0.15	8.50	1.70	57.00	29.00	14.00	0.00	0.00	0.13	0.00
WALLASEA 1	500.00	1.27	0.16	3.50	0.70	51.00	46.00	3.00	0.00	0.00	0.31	0.00
HENSE	350.00	0.20	0.48	49.00	33.00	29.00	36.00	35.00	0.00	0.00	0.13	0.00
CHARITY 1U	450.00	1.42	0.17	14.40	0.50	27.00	60.00	13.00	15.00	0.00	0.25	0.00
WICKHAM 2U	400.00	1.45	0.15	29.90	0.90	26.00	42.00	32.00	10.00	0.00	0.16	0.00

Appendix D4: Layer 3 soil data in SWAT *.sol file.

SWAT Variable Symbol												
MUID	SOL_Z3	SOL_BD3	SOL_AWC3	SOL_K3	SOL_CBN3	CLAY3	SILT3	SAND3	ROCK3	SOL_ALB3	USLE_K3	SOL_EC3
WORCESTER	600.00	1.27	0.15	2.79	1.40	55.00	37.00	8.00	0.00	0.00	0.18	0.00
EVESHAM 2	750.00	1.36	0.13	0.25	0.60	58.00	35.00	7.00	0.00	0.00	0.21	0.00
COOMBE 1	800.00	1.36	0.18	33.67	0.70	25.00	53.00	22.00	20.00	0.00	0.19	0.00
BEARSTED 2	650.00	1.30	0.18	163.17	0.60	8.00	32.00	60.00	10.00	0.00	0.18	0.00
SOUTH PETHERTON	750.00	1.28	0.19	141.88	0.60	11.00	36.00	53.00	0.00	0.00	0.18	0.00
ARDINGTON	650.00	1.47	0.13	86.96	0.70	24.00	7.00	69.00	0.00	0.00	0.12	0.00
CHARITY 1	950.00	1.37	0.17	16.50	0.30	30.00	59.00	11.00	15.00	0.00	0.26	0.00
WHIMPLE 3	600.00	1.45	0.14	1.33	0.40	44.00	44.00	12.00	10.00	0.00	0.20	0.00
OXPASTURE	750.00	1.43	0.13	0.33	0.40	50.00	39.00	11.00	0.00	0.00	0.20	0.00
CURTISDEN	1500.00	1.50	0.19	18.75	0.20	11.00	80.00	9.00	4.00	0.00	0.37	0.00
WATERSTOCK	800.00	1.45	0.14	61.83	0.80	24.00	24.00	52.00	10.00	0.00	0.16	0.00
BATCOMBE	950.00	1.43	0.12	1.00	0.50	53.00	28.00	19.00	10.00	0.00	0.15	0.00
HORNBEAM 2	550.00	1.46	0.15	23.71	0.70	29.00	40.00	31.00	10.00	0.00	0.17	0.00
BROCKHURST 1	700.00	1.44	0.13	1.71	0.40	46.00	40.00	14.00	0.00	0.00	0.19	0.00
MARTOCK	800.00	1.53	0.14	0.13	0.50	41.00	56.00	3.00	5.00	0.00	0.35	0.00
WICKHAM 2	700.00	1.39	0.14	2.13	0.60	48.00	40.00	12.00	0.00	0.00	0.19	0.00
DENCHWORTH	700.00	1.31	0.13	0.21	0.80	64.00	31.00	5.00	0.00	0.00	0.22	0.00
DUNKESWELL	800.00	1.35	0.14	4.38	0.60	48.00	40.00	12.00	10.00	0.00	0.19	0.00
FLADBURY 1	700.00	1.29	0.15	24.33	1.10	42.00	33.00	25.00	0.00	0.00	0.14	0.00
WALLASEA 1	700.00	1.25	0.15	1.50	0.70	57.00	42.00	1.00	0.00	0.00	0.34	0.00
HENSE	1500.00	1.43	0.14	135.63	0.30	15.00	12.00	73.00	5.00	0.00	0.14	0.00
CHARITY 1U	950.00	1.37	0.17	16.50	0.30	30.00	59.00	11.00	15.00	0.00	0.26	0.00
WICKHAM 2U	700.00	1.39	0.14	2.13	0.60	48.00	40.00	12.00	0.00	0.00	0.19	0.00

Appendix D5: Layer 4 soil data in SWAT *.sol file.

SWAT Variable Symbol												
MUID	SOL_Z4	SOL_BD4	SOL_AWC4	SOL_K4	SOL_CBN4	CLAY4	SILT4	SAND4	ROCK4	SOL_ALB4	USLE_K4	SOL_EC4
WORCESTER	900.00	1.41	0.14	4.38	0.60	42.00	45.00	13.00	0.00	0.00	0.20	0.00
EVESHAM 2	1500.00	1.40	0.12	0.21	0.40	58.00	34.00	8.00	0.00	0.00	0.20	0.00
COOMBE 1	1500.00	1.41	0.17	33.67	0.30	22.00	55.00	23.00	20.00	0.00	0.20	0.00
BEARSTED 2	900.00	1.45	0.16	140.29	0.20	6.00	27.00	67.00	12.00	0.00	0.18	0.00
SOUTH PETHERTON	1100.00	1.49	0.16	99.75	0.10	9.00	33.00	58.00	0.00	0.00	0.19	0.00
ARDINGTON	1000.00	1.42	0.14	140.29	0.40	14.00	12.00	74.00	0.00	0.00	0.14	0.00
CHARITY 1	1300.00	1.60	0.12	0.13	0.20	43.00	48.00	9.00	15.00	0.00	0.24	0.00
WHIMPLE 3	1000.00	1.51	0.14	3.54	0.30	35.00	50.00	15.00	0.00	0.00	0.21	0.00
OXPASTURE	1200.00	1.47	0.13	0.33	0.30	47.00	42.00	11.00	0.00	0.00	0.21	0.00
CURTISDEN	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
WATERSTOCK	1100.00	1.47	0.15	63.96	0.30	20.00	30.00	50.00	50.00	0.00	0.17	0.00
BATCOMBE	1500.00	1.62	0.08	0.04	0.10	60.00	24.00	16.00	10.00	0.00	0.15	0.00
HORNBEAM 2	1050.00	1.47	0.12	2.79	0.50	48.00	28.00	24.00	10.00	0.00	0.15	0.00
BROCKHURST 1	1000.00	1.51	0.13	0.21	0.20	45.00	48.00	7.00	0.00	0.00	0.26	0.00
MARTOCK	1300.00	1.52	0.15	1.50	0.40	32.00	59.00	9.00	5.00	0.00	0.28	0.00
WICKHAM 2	1100.00	1.49	0.12	0.08	0.20	55.00	39.00	6.00	0.00	0.00	0.24	0.00
DENCHWORTH	1350.00	1.40	0.12	0.17	0.40	58.00	36.00	6.00	0.00	0.00	0.23	0.00
DUNKESWELL	1500.00	1.47	0.09	0.04	0.20	70.00	16.00	14.00	10.00	0.00	0.13	0.00
FLADBURY 1	1000.00	1.30	0.15	27.04	1.00	42.00	29.00	29.00	0.00	0.00	0.14	0.00
WALLASEA 1	850.00	1.25	0.16	3.29	0.80	52.00	47.00	1.00	0.00	0.00	0.35	0.00
HENSE	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CHARITY 1U	1300.00	1.60	0.12	0.13	0.20	43.00	48.00	9.00	15.00	0.00	0.24	0.00
WICKHAM 2U	1100.00	1.49	0.12	0.08	0.20	55.00	39.00	6.00	0.00	0.00	0.24	0.00

Appendix D6: Layer 5 soil data in SWAT *.sol file.

SWAT Variable Symbol												
MUID	SOL_Z5	SOL_BD5	SOL_AWC5	SOL_K5	SOL_CBN5	CLAY5	SILT5	SAND5	ROCK5	SOL_ALB5	USLE_K5	SOL_EC4
WORCESTER	1500.00	1.54	0.13	1.00	0.20	40.00	43.00	17.00	0.00	0.00	0.18	0.00
EVESHAM 2	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
COOMBE 1	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
BEARSTED 2	1500.00	1.46	0.16	113.00	0.10	3.00	21.00	76.00	0.00	0.00	0.17	0.00
SOUTH PETHERTON	1500.00	1.48	0.13	131.00	0.10	14.00	10.00	76.00	0.00	0.00	0.13	0.00
ARDINGTON	1500.00	1.48	0.13	99.50	0.10	12.00	6.00	82.00	0.00	0.00	0.10	0.00
CHARITY 1	1500.00	1.48	0.14	4.40	0.10	36.00	51.00	13.00	25.00	0.00	0.22	0.00
WHIMPLE 3	1500.00	1.58	0.14	1.30	0.20	31.00	56.00	13.00	0.00	0.00	0.24	0.00
OXPASTURE	1500.00	1.49	0.14	0.40	0.30	40.00	53.00	7.00	0.00	0.00	0.28	0.00
CURTISDEN	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
WATERSTOCK	1500.00	1.47	0.15	77.30	0.10	15.00	34.00	51.00	50.00	0.00	0.18	0.00
BATCOMBE	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
HORNBEAM 2	1500.00	1.46	0.13	3.30	0.40	42.00	41.00	17.00	10.00	0.00	0.18	0.00
BROCKHURST 1	1500.00	1.53	0.14	0.30	0.20	40.00	52.00	8.00	0.00	0.00	0.26	0.00
MARTOCK	1500.00	1.59	0.15	0.40	0.20	29.00	63.00	8.00	5.00	0.00	0.30	0.00
WICKHAM 2	1500.00	1.52	0.14	0.20	0.20	41.00	58.00	1.00	0.00	0.00	0.40	0.00
DENCHWORTH	1500.00	1.44	0.12	0.10	0.30	56.00	38.00	6.00	0.00	0.00	0.24	0.00
DUNKESWELL	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
FLADBURY 1	1500.00	1.22	0.16	13.90	1.60	50.00	34.00	16.00	0.00	0.00	0.13	0.00
WALLASEA 1	1500.00	1.27	0.16	6.30	0.80	46.00	51.00	3.00	0.00	0.00	0.32	0.00
HENSE	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00	0.00
CHARITY 1U	1500.00	1.48	0.14	4.40	0.10	36.00	51.00	13.00	25.00	0.00	0.22	0.00
WICKHAM 2U	1500.00	1.52	0.14	0.20	0.20	41.00	58.00	1.00	0.00	0.00	0.40	0.00

APPENDIX E

Appendix E1: ROT1 (ww/wb/rg/rg/ww) crop management dates.

Year	Crop	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser Type	Fertiliser (kg ha^{-1})	Date
1	Winter Wheat	Fertiliser	40	N - Urea (46%)	87	10-Mar
		Fertiliser	80	N - Urea (46%)	174	10-Apr
		Fertiliser	60	N - CAN (27%)	222	10-May
		Harvest/Kill				30-Aug
	Winter barley	Tillage		Plough		07-Sep
		Fertiliser	45	P - TSP (20.7%)	217	12-Sep
		Tillage		Roller Harrow		12-Sep
		Plant				12-Sep
2		Fertiliser	45	N - Urea (46%)	98	05-Mar
		Fertiliser	110	N - Urea (46%)	239	15-Apr
		Harvest/Kill				21-Jul
	Rye grass	Tillage		Plough		30-Aug
		Tillage		Roller Harrow		01-Sep
		Plant				01-Sep
3		Fertiliser	40	N - Urea (46%)	87	20-Feb
		Fertiliser	80	N - CAN (27%)	296	07-Apr
		Fertiliser	40	P - TSP (20.7%)	193	07-Apr
		Harvest only - cut 1				21-May
		Fertiliser	100	N - Urea (46%)	217	23-May
		Fertiliser	25	P - TSP (20.7%)	120	23-May
		Harvest only - cut 2				01-Jul
		Fertiliser	80	N - Urea (46%)	173	03-Jul
		Fertiliser	15	P - TSP (20.7%)	72	03-Jul
		Harvest only - cut 3				30-Aug
4	Rye grass	Fertiliser	40	N - Urea (46%)	87	20-Feb
		Fertiliser	80	N - CAN (27%)	296	07-Apr
		Fertiliser	40	P - TSP (20.7%)	193	07-Apr
		Harvest only - cut 1				21-May
		Fertiliser	100	N - Urea (46%)	217	23-May
		Fertiliser	25	P - TSP (20.7%)	120	23-May
		Harvest only - cut 2				01-Jul
		Fertiliser	80	N - Urea (46%)	173	03-Jul
		Fertiliser	15	P - TSP (20.7%)	72	03-Jul
		Harvest only - cut 3				30-Aug
		Kill				31-Aug
	Winter wheat	Tillage		Plough		10-Sep
		Fertiliser	60	P - TSP (20.7%)	290	15-Sep
		Tillage		Roller Harrow		15-Sep
		Plant				15-Sep
5		Fertiliser	40	N - Urea (46%)	87	10-Mar
		Fertiliser	80	N - Urea (46%)	174	10-Apr
		Fertiliser	60	N - CAN (27%)	222	10-May
		Harvest/Kill				30-Aug
	Winter wheat	Tillage		Plough		10-Sep
		Fertiliser	60	P - TSP (20.7%)	290	15-Sep
		Tillage		Roller Harrow		15-Sep
		Plant				15-Sep

Key: TSP – Triple-superphosphorus (FRT_SURFACE arable = 0.01), CAN – Calcium Ammonium Nitrate

Appendix E2: ROT2 (m) crop management dates.

Year	Crop	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser Type	Fertiliser (kg ha^{-1})	Date
1	Silage maize (corn)	Tillage		Plough		08-May
		Fertiliser	85	P - TSP (20.7%)	411	15-May
		Fertiliser	15	N - Urea (46%)	55	15-May
		Tillage		Roller Harrow		15-May
		Planting				15-May
		Fertiliser	40	N - Urea (46%)	87	15-Jun
		Fertiliser	70	N - Urea (46%)	152	10-Jul
		Harvest				04-Sep
		Kill				05-Sep

Key: TSP – Triple-superphosphorus (FRT_SURFACE arable = 0.01), CAN – Calcium Ammonium Nitrate

Appendix E3: ROT3 (ww/ww/sa/rg/rg/) crop management dates.

Year	Crop	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser Type	Fertiliser (kg ha^{-1})	Date
1	Winter wheat	Fertiliser	40	N - Urea (46%)	87	10-Mar
		Fertiliser	80	N - Urea (46%)	174	10-Apr
		Fertiliser	60	N - CAN (27%)	222	10-May
		Harvest/Kill				30-Aug
		Tillage		Plough		10-Sep
		Fertiliser	60	P - TSP (20.7%)	290	15-Sep
		Tillage		Roller Harrow		15-Sep
		Plant				15-Sep
2	Winter wheat	Fertiliser	40	N - Urea (46%)	87	10-Mar
		Fertiliser	80	N - Urea (46%)	174	10-Apr
		Fertiliser	60	N - CAN (27%)	222	10-May
		Harvest/Kill				30-Aug
	Set-aside (rye grass)	Tillage		Roller Harrow		01-Sep
		Plant				01-Sep
3		Harvest - mulching				20-May
		Kill				29-Aug
	Rye grass	Tillage		Plough		30-Aug
		Tillage		Roller Harrow		01-Sep
		Plant				01-Sep
4	Rye grass	Fertiliser	40	N - Urea (46%)	87	20-Feb
		Fertiliser	80	N - CAN (46%)	296	07-Apr
		Fertiliser	40	P - TSP (20.7%)	193	07-Apr
		Harvest only - cut 1				21-May
		Fertiliser	100	N - Urea (46%)	217	23-May
		Fertiliser	25	P - TSP (20.7%)	120	23-May
		Harvest only - cut 2				01-Jul
		Fertiliser	80	N - Urea (46%)	173	03-Jul
		Fertiliser	15	P - TSP (20.7%)	72	03-Jul
		Harvest only - cut 3				30-Aug
5	Rye grass	Fertiliser	40	N - Urea (46%)	87	20-Feb
		Fertiliser	80	N - CAN (46%)	296	07-Apr
		Fertiliser	40	P - TSP (20.7%)	193	07-Apr
		Harvest only - cut 1				21-May
		Fertiliser	100	N - Urea (46%)	217	23-May
		Fertiliser	25	P - TSP (20.7%)	120	23-May
		Harvest only - cut 2				01-Jul
		Fertiliser	80	N - Urea (46%)	173	03-Jul
		Fertiliser	15	P - TSP (20.7%)	72	03-Jul
		Harvest only - cut 3				30-Aug
	Rye grass	Kill				31-Aug
	Winter wheat	Tillage		Plough		10-Sep
		Fertiliser	60	P - TSP (20.7%)	290	15-Sep
		Tillage		Roller Harrow		15-Sep
		Plant				15-Sep

Key: TSP – Triple-superphosphorus (FRT_SURFACE arable = 0.01), CAN – Calcium Ammonium Nitrate

Appendix E4: ROT4 (rg/m/wb) crop management dates.

Year	Crop	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser type	Fertiliser (kg ha^{-1})	Date
1	Rye grass	Fertiliser	40	N - Urea (46%)	87	15-Mar
		Fertiliser	20	P - TSP (20.7%)	96	15-Mar
		Grazing (30 days)		Dairy (6LU)		01-Apr
		Kill				07-May
	Maize silage	Plough		Plough		08-May
		Fertiliser	15	N - Urea (46%)	55	15-May
		Fertiliser	85	P - TSP (20.7%)	411	15-May
		Tillage		Roller Harrow		15-May
		Plant				15-May
		Fertiliser	40	N - Urea (46%)	87	15-Jun
		Fertiliser	70	N - Urea (46%)	152	10-Jul
		Harvest				03-Sep
		Kill				05-Sep
	Winter barley	Tillage		Plough		07-Sep
		Fertiliser	45	P - TSP (20.7%)	217	10-Sep
		Tillage		Roller Harrow		10-Sep
		Plant				10-Sep
2		Fertiliser	45	N - Urea (46%)	98	05-Mar
		Fertiliser	110	N - Urea (46%)	239	15-Apr
		Harvest/Kill				21-Jul
	Rye grass	Tillage		Plough		30-Aug
		Tillage		Roller Harrow		01-Sep
		Plant				01-Sep

Key: TSP – Triple-superphosphorus (FRT_SURFACE arable = 0.01)

Appendix E5: ROT5 (rg/rg/rg/rg/rg) management dates.

Year	Crop	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser type	Fertiliser (kg ha^{-1})	Date
1	Rye grass	Fertiliser	40	N - Urea (46%)	87	20-Feb
		Fertiliser	80	N - Urea (46%)	174	07-Apr
		Fertiliser	40	P - TSP (20.7%)	193	07-Apr
		Harvest only - cut 1				21-May
		Fertiliser	100	N - Urea (46%)	217	23-May
		Fertiliser	25	P - TSP (20.7%)	120	23-May
		Harvest only - cut 2				01-Jul
		Fertiliser	100	N - Urea (46%)	217	03-Jul
		Fertiliser	15	P - TSP (20.7%)	72.5	03-Jul
		Harvest only - cut 3				14-Aug
		Fertiliser	60	N - Urea (46%)	130	16-Aug
		Fertiliser	10	P - TSP (20.7%)	48	16-Aug
		Harvest only - cut 4				02-Oct
2	Rye grass	Fertiliser	40	N - Urea (46%)	87	20-Feb
		Fertiliser	80	N - Urea (46%)	174	07-Apr
		Fertiliser	40	P - TSP (20.7%)	193	07-Apr
		Harvest only - cut 1				21-May
		Fertiliser	100	N - Urea (46%)	217	23-May
		Fertiliser	25	P - TSP (20.7%)	120	23-May
3		Harvest only - cut 2				01-Jul
		Fertiliser	100	N - Urea (46%)	217	03-Jul
		Fertiliser	15	P - TSP (20.7%)	72.5	03-Jul
		Harvest only - cut 3				14-Aug
4		Fertiliser	60	N - Urea (46%)	130	16-Aug
		Fertiliser	10	P - TSP (20.7%)	48	16-Aug
		Harvest only - cut 4				02-Oct
5	Rye grass	Fertiliser	40	N - Urea (46%)	87	20-Feb
		Fertiliser	80	N - Urea (46%)	174	07-Apr
		Fertiliser	40	P - TSP (20.7%)	193	07-Apr
		Harvest only - cut 1				21-May
		Fertiliser	100	N - Urea (46%)	217	23-May
		Fertiliser	25	P - TSP (20.7%)	120	23-May
		Harvest only - cut 1				01-Jul
		Fertiliser	100	N - Urea (46%)	217	03-Jul
		Fertiliser	15	P - TSP (20.7%)	72.5	03-Jul
		Harvest only - cut 3				14-Aug
		Kill				14-Aug
		Tillage		Plough		18-Aug
		Tillage		Roller Harrow		20-Aug
		Plant				20-Aug

Key: TSP – Triple-superphosphorus (FRT_SURFACE = 0.01)

Appendix E6: PA11 and PA12 sheep grazing management dates.

PA11	Operation	Elemental fertiliser (kg ha^{-1})	Type	Amount (kg ha^{-1})	Date
Sheep grazing	Plant/ Begin growing season				01-Jan
	Fertiliser	60	N - Urea (46%)	130	15-Mar
	Fertiliser	40	P - TSP (20.7%)	193	15-Mar
	Grazing - 30 days				01-May
	Fertiliser	60	N - Urea (46%)	130	04-Jun
	Grazing - 30 days				01-Jul
	Fertiliser	50	N - Urea (46%)	108	04-Aug
	Grazing - 30 days				01-Sep

PA12	Operation	Amount elemental (kg/ha)	Type	Amount (kg/ha)	Date
Sheep grazing	Plant/ Begin growing season				01-Jan
	Fertiliser	60	N - Urea (46%)	130	15-Mar
	Fertiliser	40	P - TSP (20.7%)	193	15-Mar
	Grazing - 30 days				01-Apr
	Fertiliser	60	N - Urea (46%)	130	06-May
	Grazing - 30 days				01-Jun
	Fertiliser	50	N - Urea (46%)	108	04-Jul
	Grazing - 30 days				01-Aug

Key: TSP – Triple-superphosphorus

Appendix E7: PAS1 and PAS2 dairy cow grazing management dates.

PAS2	Operation	Amount elemental (kg ha^{-1})	Type	Amount (kg ha^{-1})	Date
Dairy cow grazing	Plant/ Begin growing season				01-Jan
	Fertiliser	60	N - Urea (46%)	130	15-Mar
	Fertiliser	40	P - STP (20.7%)	193	15-Mar
	Grazing - 30 days				01-Apr
	Fertiliser	60	N - Urea (46%)	130	06-Jun
	Grazing - 30 days				05-Jul
	Fertiliser	60	N - Urea (46%)	130	06-Aug
	Grazing - 30 days				05-Sep

PAS3	Operation	Amount elemental (kg ha^{-1})	Type	Amount (kg ha^{-1})	Date
Dairy cow grazing	Plant/ Begin growing season				01-Jan
	Fertiliser	100	N - Urea (46%)	217	15-Mar
	Fertiliser	60	P - STP (20.7%)	290	15-Mar
	Harvest-silage				15-May
	Fertiliser	100	N - Urea (46%)	217	17-May
	Grazing - 30 days				05-Jun
	Fertiliser	60	N - Urea (46%)	130	07-Jul
	Grazing - 30 days				05-Aug

Key: TSP – Triple-superphosphorus

Appendix E8: PSG4 grass management dates.

PSG4	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser type	Fertiliser (kg ha^{-1})	Date
Grass silage	Plant/ Begin growing season				01-Jan
	Fertiliser	40	N - Urea (46%)	87	01-Mar
	Fertiliser	100	N - Urea (46%)	217	10-Apr
	Fertiliser	50	P - STP (20.7%)	242	10-Apr
	Harvest only - silage cut 1				01-Jun
	Fertiliser	80	N - Urea (46%)	173	04-Jun
	Fertiliser	30	P - STP (20.7%)	145	04-Jun
	Harvest only - silage cut 2				15-Jul
	Fertiliser	50	N - Urea (46%)	109	18-Jul
	Harvest only - silage cut 3				15-Sep

Key: TSP – Triple-superphosphorus

Appendix E9: ROTE (rc/rc/ww/wb) crop management dates.

Year	Crop	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser type	Fertiliser (kg ha^{-1})	Date
1	Red clover	Fertiliser	45	P - STP (20.7%)	217	07-Apr
		Harvest - cut 1				01-Jun
		Fertiliser	25	P - STP (20.7%)	120	04-Jun
		Harvest - cut 2				20-Jul
		Harvest - cut 3				15-Sep
2	Red clover	Fertiliser	45	P - STP (20.7%)	217	07-Apr
		Harvest - cut 1				25-May
		Fertiliser	25	P - STP (20.7%)	120	28-May
		Harvest - cut 2				10-Jul
		Harvest - cut 3				04-Sep
		Kill				09-Sep
	Winter wheat	Tillage		Plough		10-Sep
		Fertiliser	50	P - STP (20.7%)	242	15-Sep
		Tillage		Roller Harrow		15-Sep
		Plant				15-Sep
3		Fertiliser	80	N - Urea (46%)	174	10-Apr
		Fertiliser	70	N - CAN (27%)	259	10-May
		Harvest/Kill				30-Aug
	Winter barley	Tillage		Plough		07-Sep
		Fertiliser	70	P	338	12-Sep
		Tillage		Roller Harrow		12-Sep
		Plant				12-Sep
4		Fertiliser	50	N - Urea (46%)	109	15-Mar
		Fertiliser	95	N - Urea (46%)	207	20-Apr
		Harvest/Kill				21-Jul
	Red clover	Tillage		Plough		01-Aug
		Fertiliser	60	P - STP (20.7%)	290	05-Aug
		Tillage		Roller Harrow		05-Aug
		Plant				05-Aug
		Harvest - cut				02-Oct

Key: TSP – Triple-superphosphorus, CAN – Calcium Ammonium Nitrate

Appendix E10: RTE5 (rg/rg/rg/rc/rc) management dates.

Year	Crop	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser type	Fertiliser (kg ha^{-1})	Date
1	Rye grass	Harvest only - cut 1				21-May
		Fertiliser	80	N - Urea (46%)	174	23-May
		Fertiliser	25	P - STP (20.7%)	120	23-May
		Harvest only - cut 2				01-Jul
		Fertiliser	80	N - Urea (46%)	174	03-Jul
		Fertiliser	15	P - STP (20.7%)	72.5	03-Jul
		Harvest only - cut 3				14-Aug
		Fertiliser	60	N - Urea (46%)	130	16-Aug
		Fertiliser	10	P - STP (20.7%)	48	16-Aug
		Harvest only - cut 4				02-Oct
		Fertiliser	40	N - Urea (46%)	87	20-Feb
2	Rye grass	Fertiliser	60	N - Urea (46%)	130	07-Apr
		Fertiliser	40	P - STP (20.7%)	193	07-Apr
		Harvest only - cut 1				21-May
		Fertiliser	90	N - Urea (46%)	196	23-May
		Fertiliser	25	P - STP (20.7%)	120	23-May
		Harvest only - cut 2				01-Jul
		Fertiliser	80	N - Urea (46%)	174	03-Jul
		Fertiliser	15	P - STP (20.7%)	72.5	03-Jul
		Harvest only - cut 3				14-Aug
		Fertiliser	60	N - Urea (46%)	130	16-Aug
		Fertiliser	10	P - STP (20.7%)	48	16-Aug
		Harvest only - cut 4				02-Oct
	Rye grass	Fertiliser	40	N - Urea (46%)	87	20-Feb
3		Fertiliser	60	N - Urea (46%)	130	07-Apr
		Fertiliser	40	P - STP (20.7%)	193	07-Apr
		Harvest only - cut 1				21-May
		Fertiliser	100	N - Urea (46%)	217	23-May
		Fertiliser	25	P - STP (20.7%)	120	23-May
		Harvest only - cut 2				01-Jul
		Fertiliser	80	N - Urea (46%)	174	03-Jul
		Fertiliser	15	P - STP (20.7%)	72.5	03-Jul
		Harvest only - cut 3				14-Aug
		Kill				15-Aug
	Red clover	Tillage		Plough		18-Aug
		Tillage		Roller Harrow		20-Aug
		Plant				20-Aug
4		Fertiliser	45	P - STP (20.7%)	217	05-Apr
		Harvest - cut 1				01-Jun
		Fertiliser	25	P - STP (20.7%)	120	08-Jun
		Harvest - cut 2				20-Jul
		Harvest - cut 3				15-Sep
5	Red clover	Fertiliser	45	P - STP (20.7%)	217	05-Apr
		Harvest - cut 1				15-Jun
		Fertiliser	25	P - STP (20.7%)	120	20-Jun
		Harvest - cut 2				20-Aug
		Kill				21-Aug
	Rye grass	Tillage		Plough		22-Aug
		Tillage		Roller Harrow		24-Aug
		Plant				24-Aug

Key: TSP – Triple-superphosphorus

Appendix E11: PG4E extensive grass management dates.

PG4E	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser type	Fertiliser (kg ha^{-1})	Date
Grass silage	Plant/ Begin growing season				01-Jan
	Fertiliser	60	N - Urea (46%)	130	10-Apr
	Fertiliser	50	P - STP (20.7%)	242	10-Apr
	Harvest only - silage cut 1				01-Jun
	Fertiliser	80	N - Urea (46%)	173	04-Jun
	Fertiliser	30	P - STP (20.7%)	145	04-Jun
	Harvest only - silage cut 2				15-Jul
	Fertiliser	30	N - Urea (46%)	65	18-Jul
	Harvest only - silage cut 3				15-Sep

Key: TSP – Triple-superphosphorus

Appendix E12: P11S and P12S extensive and sheep scenario grazing management dates.

P11S	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser type	Fertiliser (kg ha^{-1})	Date
Sheep grazing	Plant/ Begin growing season				01-Jan
	Fertiliser	30	N - Urea (46%)	65	15-Mar
	Fertiliser	20	P - STP (20.7%)	97	15-Mar
	Grazing - 30 days				01-May
	Fertiliser	30	N - Urea (46%)	65	04-Jun
	Grazing - 30 days				01-Jul
	Fertiliser	25	N - Urea (46%)	54	04-Aug
	Grazing - 30 days				01-Sep
P12S	**Operation**	**Elemental fertiliser (kg ha^{-1})**	**Fertiliser type**	**Fertiliser (kg ha^{-1})**	**Date**
Sheep grazing	Plant/ Begin growing season				01-Jan
	Fertiliser	30	N - Urea (46%)	65	15-Mar
	Fertiliser	20	P - STP (20.7%)	97	15-Mar
	Grazing - 30 days				01-Apr
	Fertiliser	30	N - Urea (46%)	65	06-May
	Grazing - 30 days				01-Jun
	Fertiliser	25	N - Urea (46%)	54	04-Jul
	Grazing - 30 days				01-Aug

Key: TSP – Triple-superphosphorus

Appendix E13: PG1S sheep scenario grass management dates.

PG1S	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser type	Fertiliser (kg ha^{-1})	Date
Hay	Plant/ Begin growing season				01-Jan
	Fertiliser	60	N - Urea (46%)	130	10-Apr
	Fertiliser	50	P - STP (20.7%)	242	10-Apr
	Harvest only - cut 1				10-Jun
	Fertiliser	80	N - Urea (46%)	173	16-Jun
	Fertiliser	30	P - STP (20.7%)	145	16-Jun
	Harvest only - cut 2				26-Jul
	Fertiliser	30	N - Urea (46%)	65	02-Aug
	Grazing - sheep (30 days)				01-Sep

Key: TSP – Triple-superphosphorus

Appendix E14: PG2S sheep scenario grass management dates.

PG2S	Operation	Elemental fertiliser (kg ha^{-1})	Fertiliser type	Fertiliser (kg ha^{-1})	Date
Hay	Plant/ Begin growing season				01-Jan
	Fertiliser	173	N - Urea (46%)	80	10-Apr
	Fertiliser	242	P - STP (20.7%)	50	10-Apr
	Harvest only - cut 1				15-Jun
	Fertiliser	195	N - Urea (46%)	90	25-Jun
	Fertiliser	145	P - STP (20.7%)	30	25-Jun
	Harvest only - cut 2				20-Aug

Key: TSP – Triple-superphosphorus

APPENDIX F

Appendix F1: Distribution of crop rotations across arable (AGRC) HRU-s.

SWAT Land Use	Soil Series	Total Area (ha)	Total Area (%)	AGRC Area (%)	Proposed crops (Findlay et al. 1984)	Rotation used in model
AGRC	Soil-431	153.06	0.38	1.87	crop	ROT2
	Soil-511f	140.21	0.35	1.71	crop	ROT4
	Soil-541B	1289.23	3.22	15.74	grass	ROT5
	Soil-571h	176.36	0.44	2.15	crop	ROT4
	Soil-571l	906.89	2.26	11.07	crop	ROT3
	Soil-572f	794.15	1.98	9.70	crop	ROT1
	Soil-572h	18.55	0.05	0.23	crop	ROT5
	Soil-572i	456.34	1.14	5.57	crop/grass	ROT4
	Soil-573a	114.84	0.29	1.40	grass	ROT5
	Soil-582a	1815.40	4.53	22.16	crop	ROT1
	Soil-711f	892.64	2.23	10.90	crop	ROT2
	Soil-712b	68.15	0.17	0.83	grass	ROT5
	Soil-714a	482.26	1.20	5.89	crop	ROT3
	Soil-813f	69.83	0.17	0.85	grass	ROT5
	Soil-871b	813.41	2.03	9.93	crop	ROT3

Appendix F2: Distribution of grass rotations across pasture (PAUK) HRU-s.

SWAT Land Use	Soil Series	Total Area (ha)	Total Area (%)	PAUK Area (%)	Proposed systems (Findlay et al. 1984)	Rotation used in model
PAUK	Soil-431	631.20	1.6	2.0	pasture (dairy/sheep)	PSG4
	Soil-541B	2339.79	5.8	7.6	pasture (sheep)	PA11
	Soil-571h	707.70	1.8	2.3	grass/pasture (dairy)	PAS2
	Soil-571l	1914.11	4.8	6.2	grass/pasture (sheep/dairy)	PAS3
	Soil-572f	7688.85	19.2	24.9	grass/pasture (dairy/sheep)	PSG4
	Soil-572h	460.16	1.1	1.5	grass/pasture (sheep/dairy)	PAS3
	Soil-572i	2158.63	5.4	7.0	grass/pasture (dairy/sheep)	PSG4
	Soil-573a	291.09	0.7	0.9	grass/pasture (dairy)	PAS3
	Soil-582a	4448.04	11.1	14.4	grass	PSG4
	Soil-711d	127.30	0.3	0.4	grass/pasture (dairy/sheep)	PA12
	Soil-711f	2833.90	7.1	9.2	grass/pasture (dairy)	PSG4
	Soil-712b	3050.22	7.6	9.9	grass/pasture (sheep)	PSG4
	Soil-714a	796.22	2.0	2.6	grass/pasture (sheep)	PA12
	Soil-813b	1421.59	3.5	4.6	grass/pasture (sheep)	PA12
	Soil-813f	53.97	0.1	0.2	grass/pasture (dairy/sheep)	PSG4
	Soil-871b	1896.18	4.7	6.2	grass/pasture (dairy/sheep)	PAS2

Appendix F3: Distribution of crop rotations across arable (AGRC) HRU-s for extensive scenarios 2.

SWAT Land Use	Soil Series	Total Area (ha)	Total Area (%)	AGRC Area (%)	Rotation used in baseline scenario	Rotation used in extensive scenario 2	Rotation sequence
AGRC	Soil-431	153.06	0.38	1.87	ROT2	ROTE	rc/ww/wb/rc
	Soil-511f	140.21	0.35	1.71	ROT4	ROTE	wb/rc/rc/ww
	Soil-541B	1289.23	3.22	15.74	ROT5	RT5E	rg/rg/rg/rc/rc
	Soil-571h	176.36	0.44	2.15	ROT4	ROTE	wb/rc/rc/ww
	Soil-571l	906.89	2.26	11.07	ROT3	ROTE	ww/wb/rc/rc
	Soil-572f	794.15	1.98	9.70	ROT1	ROTE	rc/rc/ww/wb
	Soil-572h	18.55	0.05	0.23	ROT5	RT5E	rg/rg/rg/rc/rc
	Soil-572i	456.34	1.14	5.57	ROT4	ROTE	wb/rc/rc/ww
	Soil-573a	114.84	0.29	1.40	ROT5	RT5E	rg/rg/rg/rc/rc
	Soil-582a	1815.40	4.53	22.16	ROT1	ROTE	rc/rc/ww/wb
	Soil-711f	892.64	2.23	10.90	ROT2	ROTE	rc/ww/wb/rc
	Soil-712b	68.15	0.17	0.83	ROT5	RT5E	rg/rg/rg/rc/rc
	Soil-714a	482.26	1.20	5.89	ROT3	ROTE	ww/wb/rc/rc
	Soil-813f	69.83	0.17	0.85	ROT5	RT5E	rg/rg/rg/rc/rc

Appendix F4: Distribution of rotations across arable (AGRC) HRU-s for sheep scenario 3.

SWAT Land Use	Soil Series	Total Area (ha)	Total Area (%)	AGRC Area (%)	Rotation used in baseline scenario	Rotation used in extensive scenario 2	Rotation sequence
AGRC	Soil-431	153.06	0.38	1.87	ROT2	ROTE	rc/ww/wb/rc
	Soil-511f	140.21	0.35	1.71	ROT4	ROTE	wb/rc/rc/ww
	Soil-541B	1289.23	3.22	15.74	ROT5	PG1S	cut1/cut2/gazing
	Soil-571h	176.36	0.44	2.15	ROT4	ROTE	wb/rc/rc/ww
	Soil-571l	906.89	2.26	11.07	ROT3	ROTE	ww/wb/rc/rc
	Soil-572f	794.15	1.98	9.70	ROT1	ROTE	rc/rc/ww/wb
	Soil-572h	18.55	0.05	0.23	ROT5	PG1S	cut1/cut2/gazing
	Soil-572i	456.34	1.14	5.57	ROT4	ROTE	wb/rc/rc/ww
	Soil-573a	114.84	0.29	1.40	ROT5	PG1S	cut1/cut2/gazing
	Soil-582a	1815.40	4.53	22.16	ROT1	ROTE	rc/rc/ww/wb
	Soil-711f	892.64	2.23	10.90	ROT2	ROTE	rc/ww/wb/rc
	Soil-712b	68.15	0.17	0.83	ROT5	PG1S	cut1/cut2/gazing
	Soil-714a	482.26	1.20	5.89	ROT3	ROTE	ww/wb/rc/rc
	Soil-813f	69.83	0.17	0.85	ROT5	PG1S	cut1/cut2/gazing

APPENDIX G

Appendix G1: Fertiliser information in SWAT fert.dat file

SWAT Variable Symbol							
FERTNM	FMINN	FMINP	FORGN	FORGP	FNH3N	FERTNAME	MANURE
UREA	0.460	0.000	0.000	0.000	1.000	Urea	FALSE
CAN	0.270	0.000	0.000	0.000	0.000	Calcium Ammonium Nitrate	FALSE
TSPH	0.000	0.207	0.000	0.000	0.000	Triple Superphosphate	FALSE
DAIRY-FR	0.007	0.005	0.031	0.003	0.990	Dairy Fresh Manure	TRUE
SHEEP-FR	0.014	0.003	0.024	0.005	0.990	Sheep Fresh Manure	TRUE

APPENDIX H

Appendix H1: Water Abstraction data input in SWAT *.wus file

	SWAT Abstraction Source							
	Deep aquifer (10^4 m^3 day^{-1})		Reach (10^4 m^3 day^{-1})		Shallow aquifer (10^4 m^3 day^{-1})		Total (10^4 m^3 day^{-1})	
Sub-catchment	EA	80%	EA	80%	EA	80%	EA	80%
1	-	-	-	-	0.1	0.1	0.1	0.1
3	-	-	-	-	0.4	0.3	0.4	0.3
5	-	-	0.1	0.1	-	-	0.1	0.1
6	-	-	2.6	2.1	0.3	0.2	2.9	2.3
13	-	-	0.6	0.5	-	-	0.6	0.5
15	-	-	0.4	0.3	-	-	0.4	0.3
16	-	-	0.2	0.1	0.2	0.1	0.3	0.3
18	-	-	-	-	0.3	0.3	0.3	0.3
22	-	-	0.1	0.1	-	-	0.1	0.1
24	-	-	3.6	2.8	-	-	3.6	2.8
Grand Total	**0.0**	**0.0**	**7.5**	**6.0**	**1.3**	**1.0**	**8.8**	**7.0**

APPENDIX I

Appendix I1: Point source discharge input data used in SWAT without phosphorus stripping introduction.

	SWAT Variable Symbol										
Sub-catchment	FLOCNST (m^3)	SEDCNST (tons)	ORGNCNST (kg N)	ORGPCNST (kg P)	NO3CNST (kg N)	NH3CNST (kg N)	NO2CNST (kg N)	MINPCNST (kg P)	CBODCNST (kg CBOD)	DISOXCNST (kg O_2)	CHLACNST (kg chla)
1	376.610	0.006	1.043	0.244	6.214	0.919	0.143	1.612	1.102	2.245	0.003
2	9.465	0.000	0.026	0.006	0.156	0.023	0.004	0.041	0.027	0.056	0.000
3	12.645	0.000	0.035	0.008	0.209	0.031	0.005	0.054	0.035	0.075	0.000
4	2382.635	0.037	6.600	1.549	39.313	5.814	0.905	10.198	6.687	14.201	0.017
5	350.890	0.002	0.125	0.028	0.388	0.137	0.088	0.068	0.663	1.068	0.000
6	11620.690	0.072	4.192	0.925	13.130	4.585	2.930	2.312	22.187	35.434	0.001
7	1.640	0.000	0.005	0.001	0.027	0.004	0.001	0.007	0.005	0.010	0.000
8	0.500	0.000	0.001	0.000	0.008	0.001	0.000	0.002	0.001	0.003	0.000
9	328.515	0.006	0.855	0.233	5.016	0.819	0.158	1.143	2.562	1.937	0.002
10	10.850	0.000	0.030	0.007	0.179	0.026	0.004	0.046	0.030	0.065	0.000
11	3311.860	0.074	16.954	1.822	85.139	32.416	1.073	6.918	26.686	21.861	0.024
12	1.500	0.000	0.004	0.001	0.025	0.004	0.001	0.006	0.004	0.009	0.000
13	39.710	0.001	0.110	0.026	0.655	0.097	0.015	0.170	0.111	0.237	0.000
14	112.170	0.002	0.311	0.073	1.851	0.274	0.043	0.480	0.600	0.669	0.001
15	4960.400	0.078	13.305	8.838	78.667	12.265	1.897	50.652	25.638	28.742	0.036
16	99.090	0.002	0.274	0.064	1.635	0.242	0.038	0.424	0.278	0.591	0.001
17	97.250	0.001	0.269	0.069	1.605	0.237	0.037	0.584	0.380	0.580	0.001
18	237.665	0.004	0.658	0.955	3.921	0.580	0.090	5.472	0.667	1.416	0.002
19	101.190	0.001	0.280	0.066	1.670	0.247	0.038	0.433	0.363	0.603	0.001
20	7.485	0.000	0.021	0.005	0.124	0.018	0.003	0.032	0.021	0.045	0.000
21	2488.955	0.015	0.809	0.177	2.243	0.906	0.623	0.344	4.671	7.482	0.000
22	114.635	0.001	0.301	0.071	1.795	0.265	0.041	0.466	0.305	0.648	0.001
23	49.375	0.001	0.137	0.032	0.815	0.120	0.019	0.211	0.139	0.294	0.000
24	49.850	0.001	0.138	0.033	0.823	0.122	0.019	0.211	0.140	0.297	0.000
25	1314.100	0.025	3.079	0.829	19.173	1.996	0.397	5.535	10.725	9.042	0.009
Total	28079.675	0.326	49.564	16.061	264.780	62.149	8.573	87.424	104.029	127.609	0.099

Appendix I2: Point source discharge input data used in SWAT with phosphorus stripping introduction.

	SWAT Variable Symbol										
Sub-catchment	FLOCNST (m^3)	SEDCNST (tons)	ORGNCNST (kg N)	ORGPCNST (kg P)	NO3CNST (kg N)	NH3CNST (kg N)	NO2CNST (kg N)	MINPCNST (kg P)	CBODCNST (kg CBOD)	DISOXCNST (kg O_2)	CHLACNST (kg chla)
1	376.610	0.006	0.244	1.612	1.043	6.214	0.919	0.143	1.102	2.245	36.000
2	9.465	0.000	0.006	0.041	0.026	0.156	0.023	0.004	0.027	0.056	7.000
3	12.645	0.000	0.008	0.054	0.035	0.209	0.031	0.005	0.035	0.075	14.000
4	2382.635	0.037	1.549	4.894	6.600	39.313	5.814	0.905	6.687	14.201	47.000
5	350.890	0.002	0.028	0.068	0.125	0.388	0.137	0.088	0.663	1.068	5.000
6	11620.690	0.072	0.925	2.312	4.192	13.130	4.585	2.930	22.187	35.434	22.000
7	1.640	0.000	0.001	0.007	0.005	0.027	0.004	0.001	0.005	0.010	3.000
8	0.500	0.000	0.000	0.002	0.001	0.008	0.001	0.000	0.001	0.003	1.000
9	328.515	0.006	0.233	1.143	0.855	5.016	0.819	0.158	2.562	1.937	11.000
10	10.850	0.000	0.007	0.046	0.030	0.179	0.026	0.004	0.030	0.065	4.000
11	3311.860	0.074	1.822	6.918	16.954	85.139	32.416	1.073	26.686	21.861	4.000
12	1.500	0.000	0.001	0.006	0.004	0.025	0.004	0.001	0.004	0.009	3.000
13	39.710	0.001	0.026	0.170	0.110	0.655	0.097	0.015	0.111	0.237	30.000
14	112.170	0.002	0.073	0.480	0.311	1.851	0.274	0.043	0.600	0.669	12.000
15	4960.400	0.078	1.571	8.509	13.305	78.667	12.265	1.897	25.638	28.742	23.000
16	99.090	0.002	0.064	0.424	0.274	1.635	0.242	0.038	0.278	0.591	24.000
17	97.250	0.001	0.069	0.584	0.269	1.605	0.237	0.037	0.380	0.580	14.000
18	237.665	0.004	0.186	1.116	0.658	3.921	0.580	0.090	0.667	1.416	15.000
19	101.190	0.001	0.066	0.433	0.280	1.670	0.247	0.038	0.363	0.603	5.000
20	7.485	0.000	0.005	0.032	0.021	0.124	0.018	0.003	0.021	0.045	9.000
21	2488.955	0.015	0.177	0.344	0.809	2.243	0.906	0.623	4.671	7.482	8.000
22	121.835	0.001	0.075	0.496	0.321	1.914	0.283	0.044	0.326	0.691	38.000
23	49.375	0.001	0.032	0.211	0.137	0.815	0.120	0.019	0.139	0.294	29.000
24	49.850	0.001	0.033	0.211	0.138	0.823	0.122	0.019	0.140	0.297	19.000
25	1314.100	0.025	0.829	2.720	3.079	19.173	1.996	0.397	10.725	9.042	17.000
Grand Total	28086.875	0.326	8.030	32.836	49.584	264.898	62.167	8.575	104.049	127.652	400.000

Appendix I3: Point source discharge data used for recalculation of SWAT input data.

Discharge	Sediment (mg l^{-1})	Organic N (mg l^{-1})	Organic P (mg l^{-1})	Nitrate (mg l^{-1})	Ammonia (mg l^{-1})	Nitrite (mg l^{-1})	Mineral P (mg l^{-1})	CBOD (mg l^{-1})	DO (mg l^{-1})	Chlorophyll a (mg l^{-1})
Until the end of 2001.										
Fish farms	6.000	0.320	0.070	0.870	0.360	0.250	0.130	1.875	3.000	0.000
Axminster STW	22.350	5.120	0.550	25.711	9.791	0.324	2.088	8.060	6.601	0.007
Broadwindsor STW	9.575	2.770	0.648	16.500	2.440	0.380	4.280	3.290	5.960	0.007
Buckland St Mary STW	7.321	2.770	0.650	16.500	2.440	0.380	4.280	2.807	5.960	0.007
Colyton & Colyford STW	18.990	2.330	0.630	14.530	1.490	0.300	4.210	8.330	6.910	0.007
Dalwood pumping st.	15.330	2.770	0.678	16.500	2.440	0.380	4.208	2.807	5.960	0.007
Drimpton STW	19.415	2.770	0.648	16.500	2.440	0.380	4.280	2.688	5.960	0.007
Farway STW	15.330	2.770	0.650	16.500	2.440	0.380	4.280	2.807	5.960	0.007
Hawkchurch STW	9.602	2.770	0.717	16.500	2.440	0.380	6.235	4.051	5.960	0.007
Musbury & Whitford stw	17.217	2.598	0.710	15.237	2.495	0.484	3.459	7.932	5.896	0.007
Offwell STW	19.143	2.770	0.648	16.500	2.440	0.380	4.280	5.682	5.960	0.007
Tatworth STW	17.931	1.951	0.627	10.513	2.744	0.403	3.807	6.327	4.412	0.007
Thorncombe STW	12.130	2.770	0.648	16.500	2.440	0.380	4.280	3.612	5.960	0.007
Winsham STW	16.514	2.770	0.648	16.500	2.440	0.380	7.238	8.638	5.960	0.007
Yarcombe stw	5.132	2.770	0.650	16.500	2.440	0.380	4.280	2.807	5.960	0.007
Chard Junction Quarry	15.330	2.770	0.004	16.500	2.440	0.380	0.020	5.870	5.960	0.007
St Ivel Ltd.	15.330	2.770	6.470	16.500	2.440	0.380	36.680	2.807	5.960	0.007
Lubborn Cheese Ltd.	15.330	2.770	6.470	16.500	2.440	0.380	36.680	2.807	5.960	0.007
Dairycrest Creamery	15.330	2.770	6.470	16.500	2.440	0.380	36.680	2.807	5.960	0.007
Discharges with no data	15.330	2.770	0.650	16.500	2.440	0.380	4.280	2.807	5.960	0.007
Discharges with changed phosphorus levels after stripping introduction in 2002.										
Colyton & Colyford STW	18.990	2.330	0.630	14.530	1.490	0.300	**2.000**	8.330	6.910	0.007
Horslears P.S.	15.330	2.770	0.650	16.500	2.440	0.380	**2.000**	2.807	5.960	0.007
Tatworth STW	17.931	1.951	0.627	10.513	2.744	0.403	**2.000**	6.327	4.412	0.007
St Ivel Ltd.	15.330	2.770	**0.880**	16.500	2.440	0.380	**5.000**	2.807	5.960	0.007
Lubborn Cheese Ltd.	15.330	2.770	**0.880**	16.500	2.440	0.380	**5.000**	2.807	5.960	0.007
Dairycrest Creamery	15.330	2.770	**0.880**	16.500	2.440	0.380	**5.000**	2.807	5.960	0.007

APPENDIX J

Appendix J1: Plant growth parameters in SWAT database used in the model.

SWAT Parameter												
CPNM	**IDC**	**CROPNAME**	**BIO_E**	**HVSTI**	**BLAI**	**FRGRW1**	**LAIMX1**	**FRGRW2**	**LAIMX2**	**DLAI**	**CHTMX**	**RDMX**
FRST	7	Forest-Mixed	15.00	0.76	5.00	0.05	0.05	0.40	0.95	0.99	6.00	3.50
FRSD	7	Forest-Deciduous	15.00	0.76	5.00	0.05	0.05	0.40	0.95	0.99	6.00	3.50
FRSE	7	Forest-Evergreen	15.00	0.76	5.00	0.15	0.70	0.25	0.99	0.99	10.00	3.50
CSIL	4	Corn Silage	39.00	0.90	4.00	0.20	0.15	0.40	0.95	0.70	2.50	0.50
WWHT	5	Winter Wheat	30.00	0.40	4.00	0.35	0.10	0.65	0.90	0.80	0.90	0.20
WIBA	5	Winter Barley	30.00	0.54	5.00	0.35	0.10	0.60	0.90	0.70	0.90	0.20
RYEG	5	Italian Ryegrass	30.00	0.99	4.00	0.30	0.25	0.50	0.95	0.60	0.80	0.30
CLVR	2	Red Clover	25.00	0.90	4.00	0.15	0.01	0.50	0.95	0.75	0.75	1.50
PA11	6	Pasture11-sheep	30.00	0.90	4.00	0.05	0.05	0.49	0.95	0.99	0.50	0.30
PA12	6	Pasture12-sheep	30.00	0.90	4.00	0.05	0.05	0.49	0.95	0.99	0.50	0.30
PAS2	6	Pasture2-dairy	30.00	0.90	4.00	0.05	0.05	0.49	0.95	0.99	0.50	0.30
PAS3	6	Pasture3-dairy	30.00	0.90	4.00	0.05	0.05	0.49	0.95	0.99	1.00	0.30
PSG4	6	Pasture (silage-hay)	30.00	0.99	4.00	0.05	0.05	0.49	0.95	0.99	1.00	0.30
MAGR	6	PastureWildelife	30.00	0.90	4.00	0.05	0.05	0.49	0.95	0.99	0.50	0.30

CPNM	**IDC**	**CROPNAME**	**T_OPT**	**T_BASE**	**CNYLD**	**CPYLD**	**BN1**	**BN2**	**BN3**	**BP1**	**BP2**	**BP3**
FRST	7	Forest-Mixed	30.00	10.00	0.0015	0.0003	0.0060	0.0020	0.0015	0.0007	0.0004	0.0003
FRSD	7	Forest-Deciduous	15.00	5.00	0.0015	0.0003	0.0060	0.0020	0.0015	0.0007	0.0004	0.0003
FRSE	7	Forest-Evergreen	15.00	0.00	0.0015	0.0003	0.0060	0.0020	0.0015	0.0007	0.0004	0.0003
CSIL	4	Corn Silage	18.00	4.00	0.0140	0.0016	0.0470	0.0177	0.0138	0.0048	0.0018	0.0014
WWHT	5	Winter Wheat	18.00	0.00	0.0250	0.0022	0.0663	0.0255	0.0148	0.0053	0.0020	0.0012
WIBA	5	Winter Barley	18.00	0.00	0.0250	0.0022	0.0663	0.0255	0.0148	0.0053	0.0020	0.0012
RYEG	5	Italian Ryegrass	15.00	0.00	0.0220	0.0028	0.0660	0.0254	0.0147	0.0105	0.0040	0.0024
CLVR	2	Red Clover	15.00	1.00	0.0650	0.0040	0.0650	0.0280	0.0243	0.0060	0.0024	0.0024
PA11	6	Pasture11-sheep	15.00	0.00	0.0234	0.0033	0.0570	0.0215	0.0120	0.0099	0.0032	0.0019
PA12	6	Pasture12-sheep	15.00	0.00	0.0234	0.0033	0.0570	0.0215	0.0120	0.0099	0.0032	0.0019
PAS2	6	Pasture2-dairy	15.00	0.00	0.0234	0.0033	0.0570	0.0215	0.0120	0.0099	0.0032	0.0019
PAS3	6	Pasture3-dairy	15.00	0.00	0.0234	0.0033	0.0570	0.0215	0.0120	0.0099	0.0032	0.0019
PSG4	6	Pasture (silage-hay)	15.00	0.00	0.0234	0.0033	0.0570	0.0215	0.0120	0.0099	0.0032	0.0019
MAGR	6	PastureWildelife	15.00	0.00	0.0234	0.0033	0.0570	0.0215	0.0120	0.0099	0.0032	0.0019

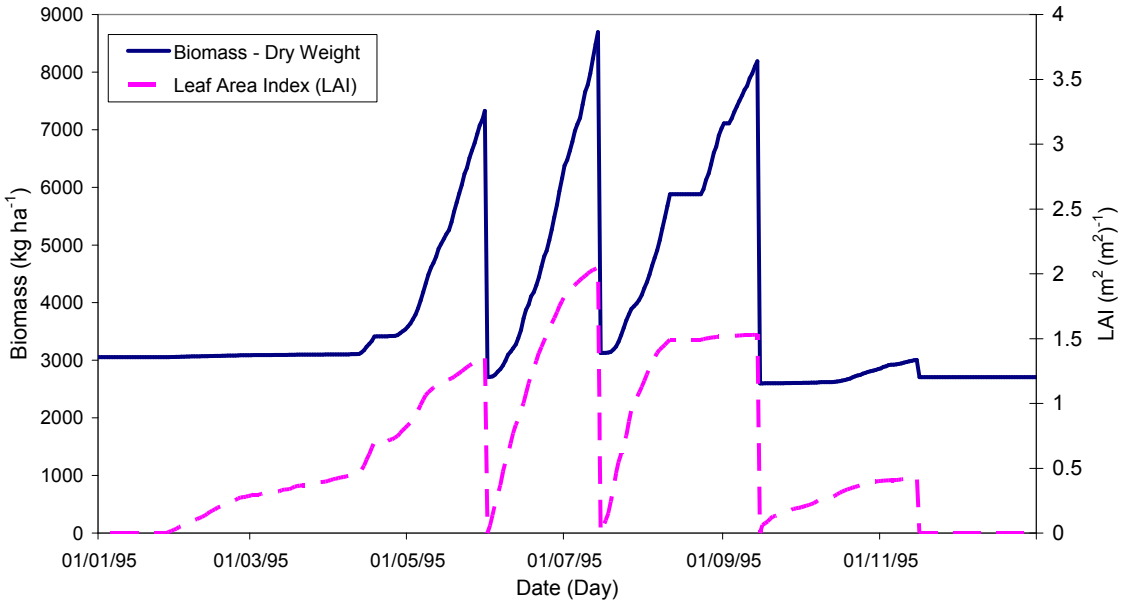
Appendix J2: Plant growth parameters in SWAT database used in the model.

SWAT Parameter												
CPNM	**IDC**	**CROPNAME**	**WSYF**	**USLE_C**	**GSI**	**VPDFR**	**FRGMAX**	**WAVP**	**CO2HI**	**BIOEHI**	**RSDCO_PL**	**OV_N**
FRST	7	Forest-Mixed	0.010	0.001	0.002	4.000	0.750	8.000	660.000	16.000	0.050	0.10
FRSD	7	Forest-Deciduous	0.010	0.001	0.002	4.000	0.750	8.000	660.000	16.000	0.050	0.60
FRSE	7	Forest-Evergreen	0.600	0.001	0.002	4.000	0.750	8.000	660.000	16.000	0.050	0.60
CSIL	4	Corn Silage	0.900	0.200	0.007	4.000	0.750	7.200	660.000	45.000	0.050	0.14
WWHT	5	Winter Wheat	0.200	0.030	0.006	4.000	0.750	6.000	660.000	39.000	0.050	0.14
WIBA	5	Winter Barley	0.200	0.030	0.006	4.000	0.750	7.000	660.000	45.000	0.050	0.14
RYEG	5	Italian Ryegrass	0.900	0.030	0.005	4.000	0.750	6.000	660.000	39.000	0.050	0.10
CLVR	2	Red Clover	0.900	0.003	0.006	4.000	0.750	10.000	660.000	30.000	0.050	0.06
PA11	6	Pasture11-sheep	0.900	0.003	0.005	4.000	0.750	8.000	660.000	36.000	0.050	0.24
PA12	6	Pasture12-sheep	0.900	0.003	0.005	4.000	0.750	8.000	660.000	36.000	0.050	0.24
PAS2	6	Pasture2-dairy	0.900	0.003	0.005	4.000	0.750	8.000	660.000	36.000	0.050	0.24
PAS3	6	Pasture3-dairy	0.900	0.003	0.005	4.000	0.750	8.000	660.000	36.000	0.050	0.24
PSG4	6	Pasture (silage-hay)	0.900	0.003	0.005	4.000	0.750	8.000	660.000	39.000	0.050	0.30
MAGR	6	Pasture Wildelife	0.900	0.003	0.005	4.000	0.750	8.000	660.000	36.000	0.050	0.24

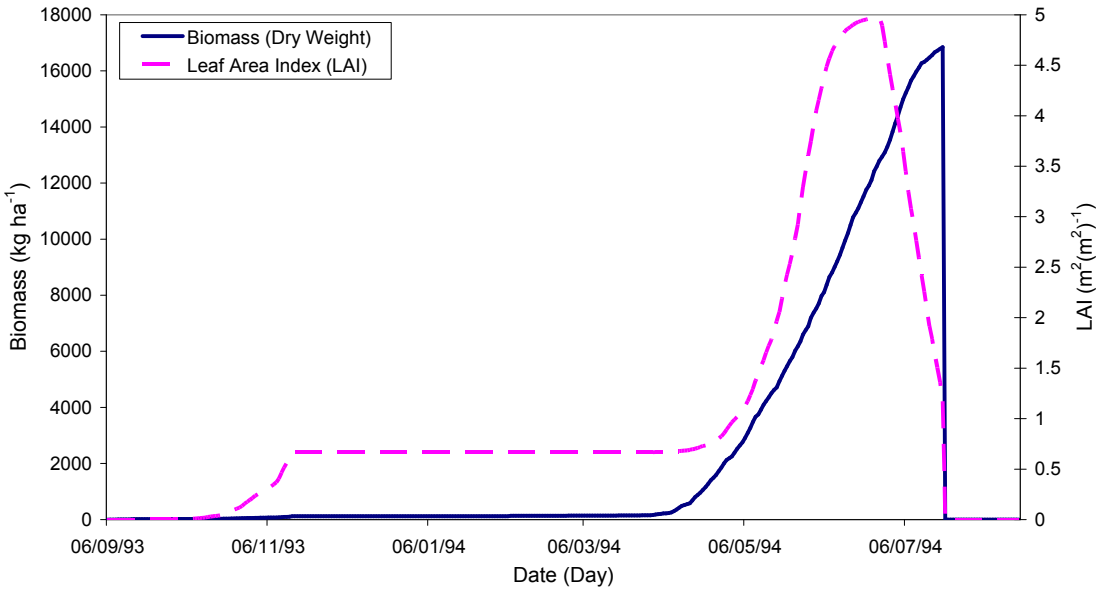
CPNM	**IDC**	**CROPNAME**	**CN2A**	**CN2B**	**CN2C**	**CN2D**	**FERTFIELD**	**ALAI_MIN**	**BIO_LEAF**
FRST	7	Forest-Mixed	25.00	55.00	70.00	77.00	FALSE	0.75	0.30
FRSD	7	Forest-Deciduous	25.00	55.00	70.00	77.00	FALSE	0.75	0.30
FRSE	7	Forest-Evergreen	25.00	55.00	70.00	77.00	FALSE	0.75	0.30
CSIL	4	Corn Silage	67.00	77.00	83.00	87.00	TRUE	0.00	0.00
WWHT	5	Winter Wheat	62.00	73.00	81.00	84.00	TRUE	0.00	0.00
WIBA	5	Winter Barley	63.00	75.00	83.00	87.00	TRUE	0.00	0.00
RYEG	5	Italian Ryegrass	31.00	59.00	72.00	79.00	TRUE	0.00	0.00
CLVR	2	Red Clover	31.00	59.00	72.00	79.00	TRUE	0.00	0.00
PA11	6	Pasture11-sheep	49.00	69.00	79.00	84.00	TRUE	0.00	0.00
PA12	6	Pasture12-sheep	49.00	69.00	79.00	84.00	TRUE	0.00	0.00
PAS2	6	Pasture2-dairy	49.00	69.00	79.00	84.00	TRUE	0.00	0.00
PAS3	6	Pasture3-dairy	49.00	69.00	79.00	84.00	TRUE	0.00	0.00
PSG4	6	Pasture (silage-hay)	30.00	58.00	71.00	78.00	TRUE	0.00	0.00
MAGR	6	Pasture Wildelife	39.00	61.00	74.00	80.00	TRUE	0.00	0.00

APPENDIX K

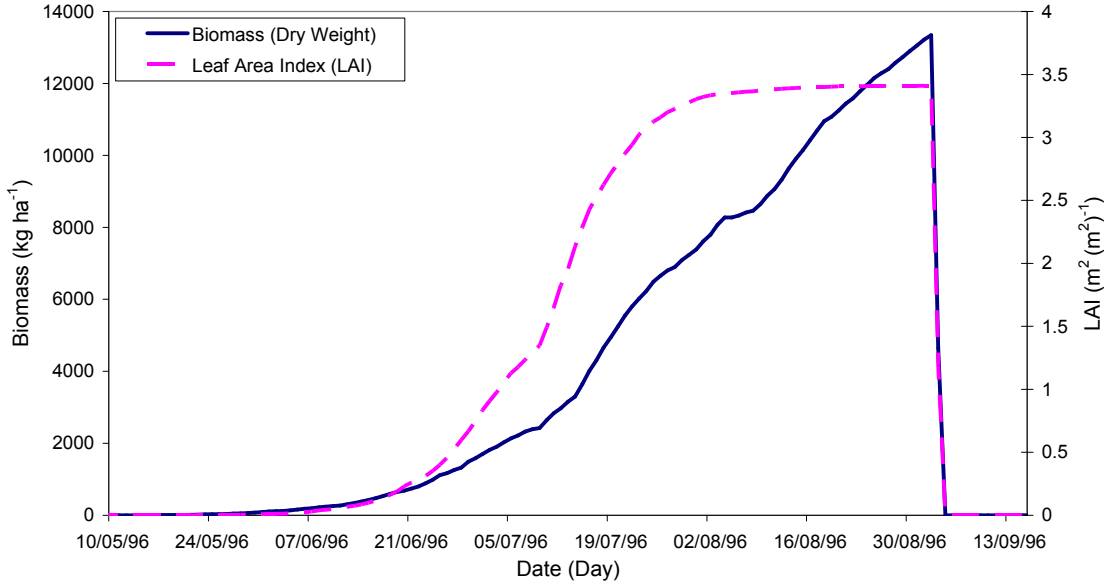
Appendix K1: Modelled plant growth for permanent grass in three cut grass silage system - PSG4 (Sub-catchment 1, HRU 1).



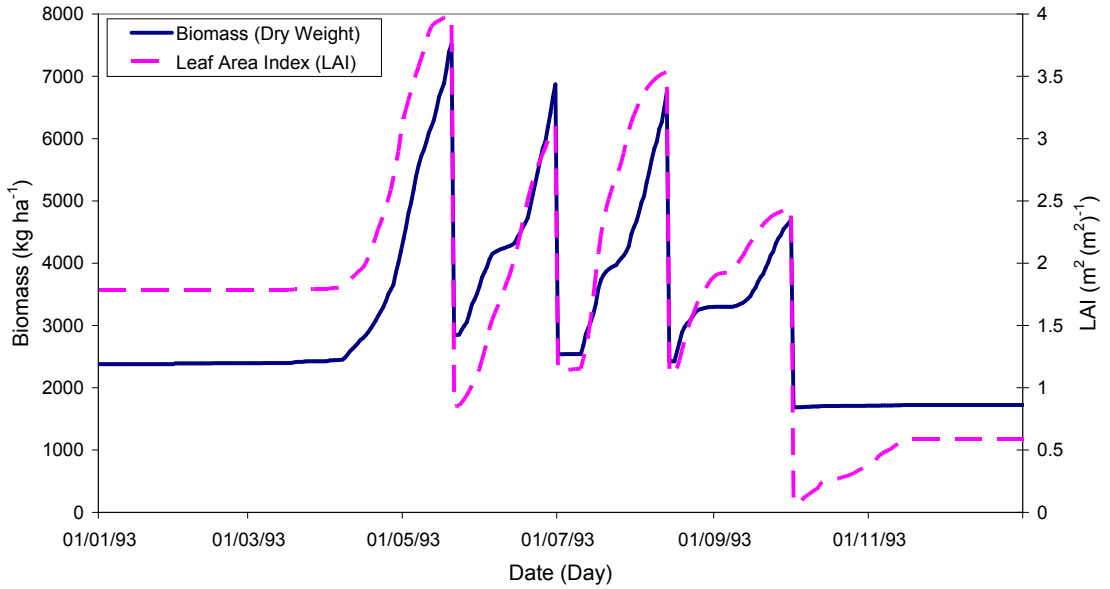
Appendix K2: Modelled plant growth for winter barley – ROT4 (Sub-catchment 1, HRU 7).



Appendix K3: Modelled plant growth for silage maize – ROT2 (Sub-catchment 1, HRU 4).

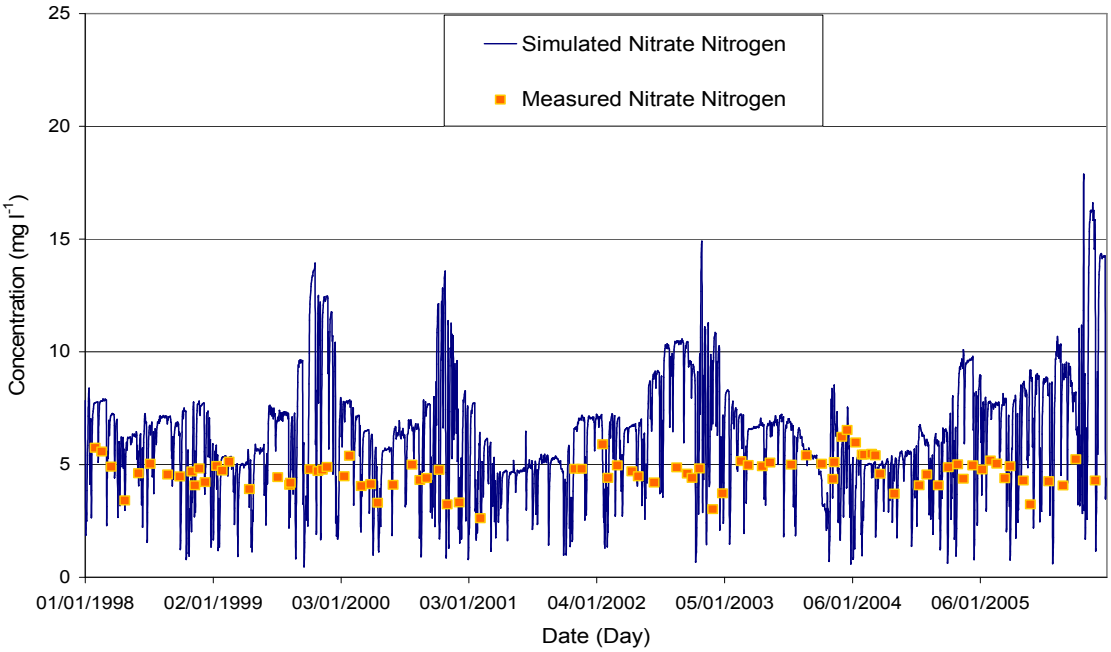


Appendix K4: Modelled plant growth for rye grass – ROT5 (Sub-catchment 5, HRU 24).

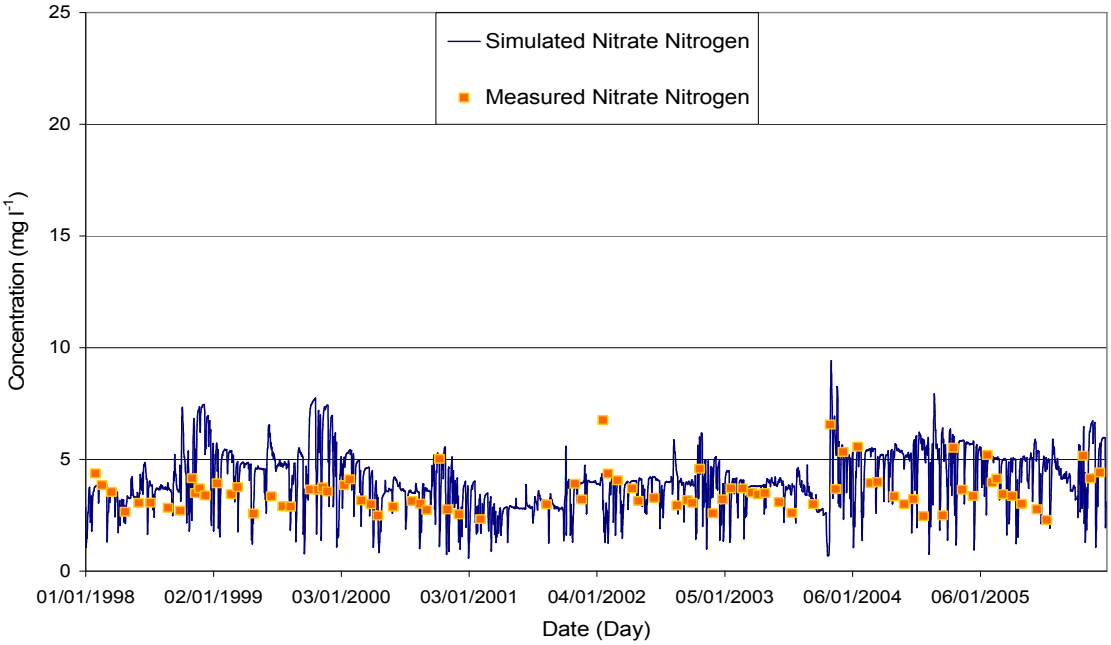


APPENDIX L

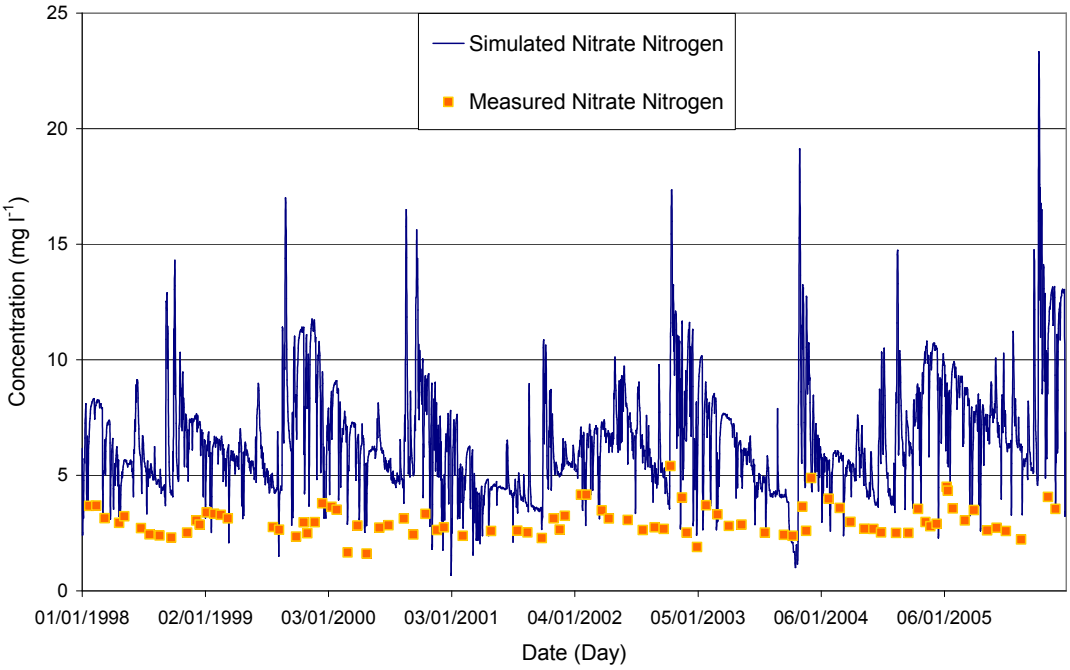
Appendix L1: Comparison between daily simulated (SWAT) and measured (EA) concentrations of nitrate nitrogen (NO_3-N) for the Slymlakes sampling point during the validation period (1998 – 2005).



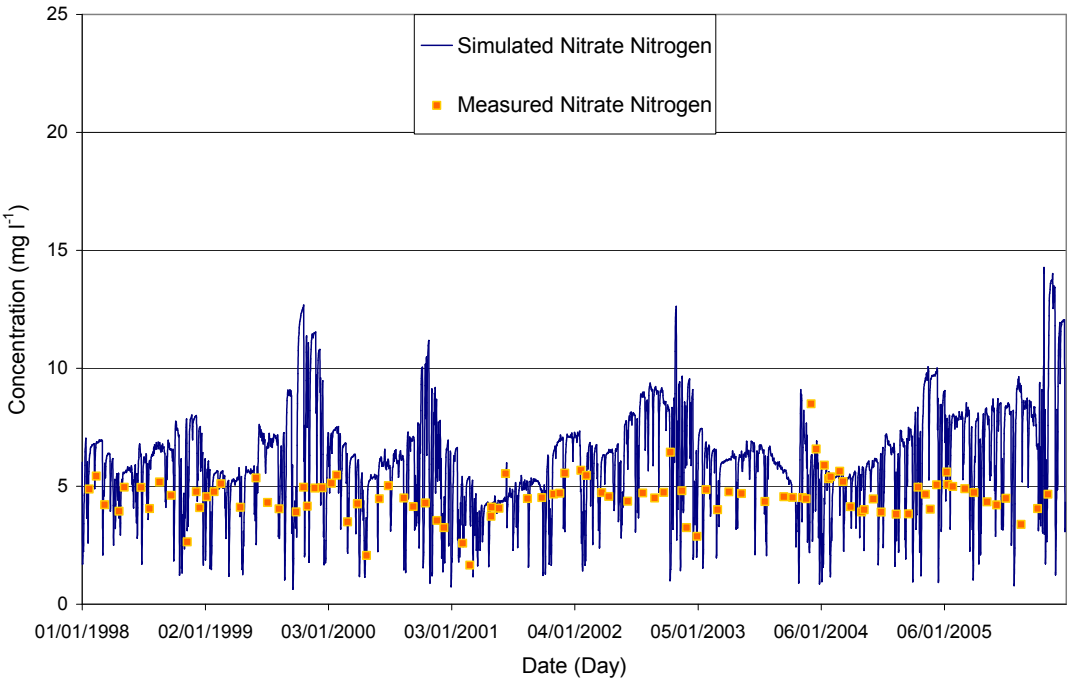
Appendix L2: Comparison between daily simulated (SWAT) and measured (EA) concentrations of nitrate nitrogen (NO_3-N) for the Newenham Abbey sampling point during the validation period (1998 – 2005).



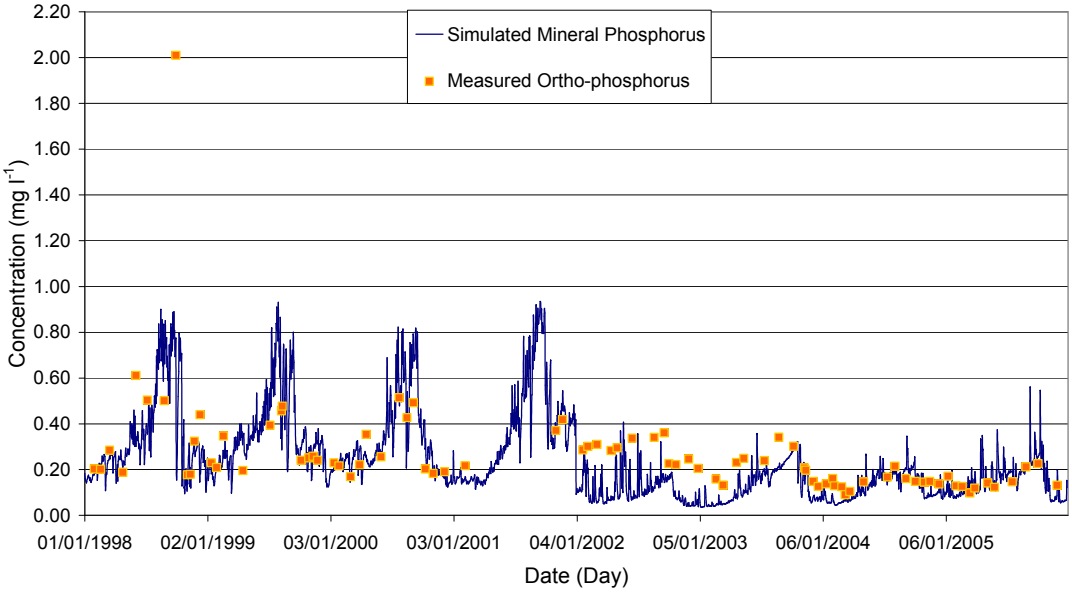
Appendix L3: Comparison between daily simulated (SWAT) and measured (EA) concentrations of nitrate nitrogen (NO_3-N) for the Colyford sampling point during the validation period (1998 – 2005).



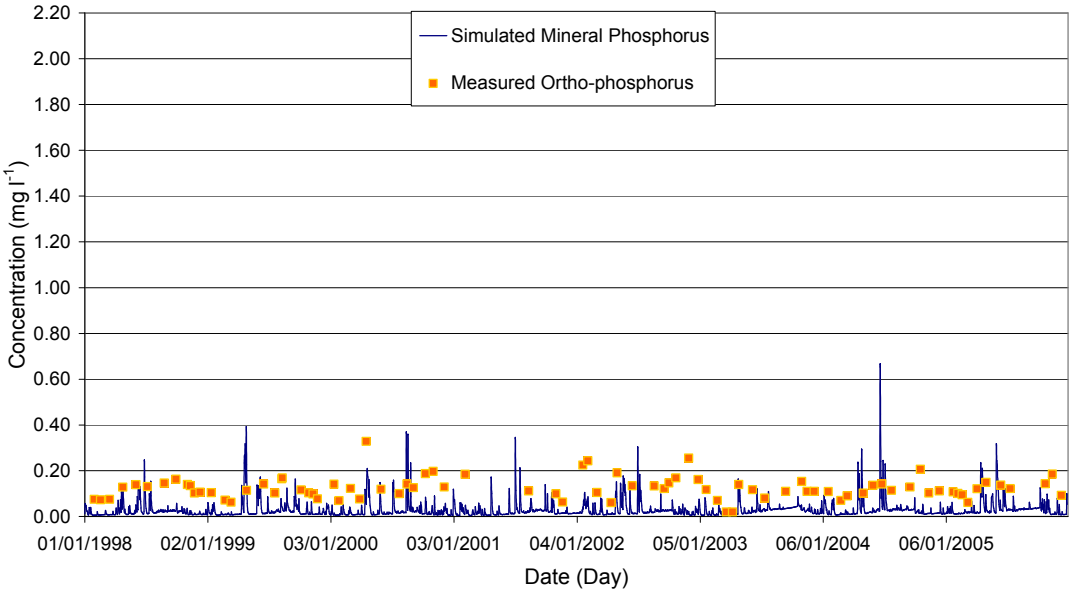
Appendix L4: Comparison between daily simulated (SWAT) and measured (EA) concentrations of nitrate nitrogen (NO_3-N) for the Axe Bridge sampling point during the validation period (1998 – 2005).



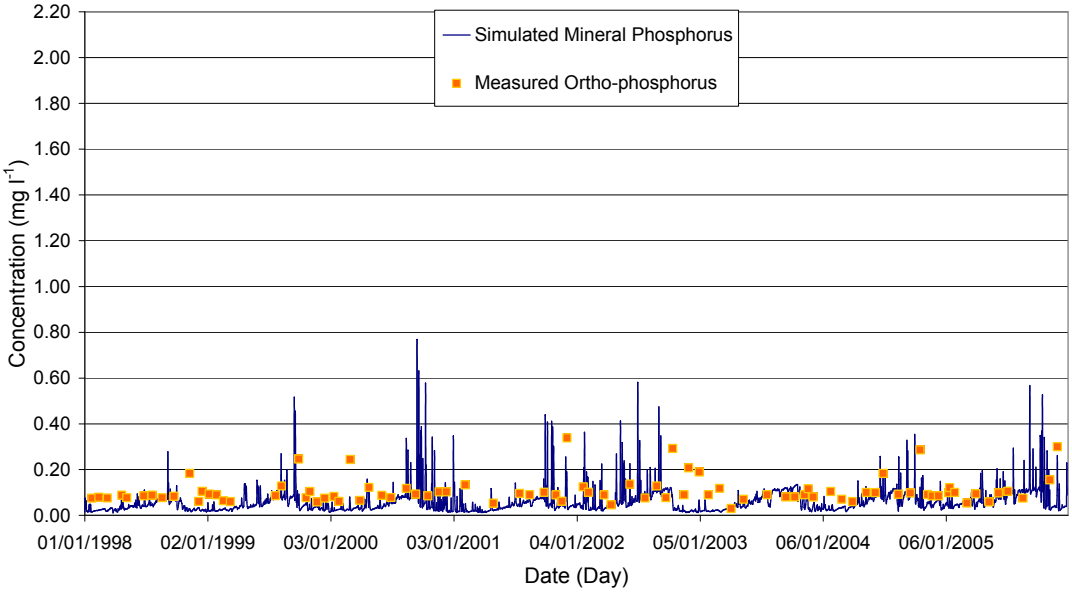
Appendix L5: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the Slymlakes sampling point during the validation period (1998 – 2005).



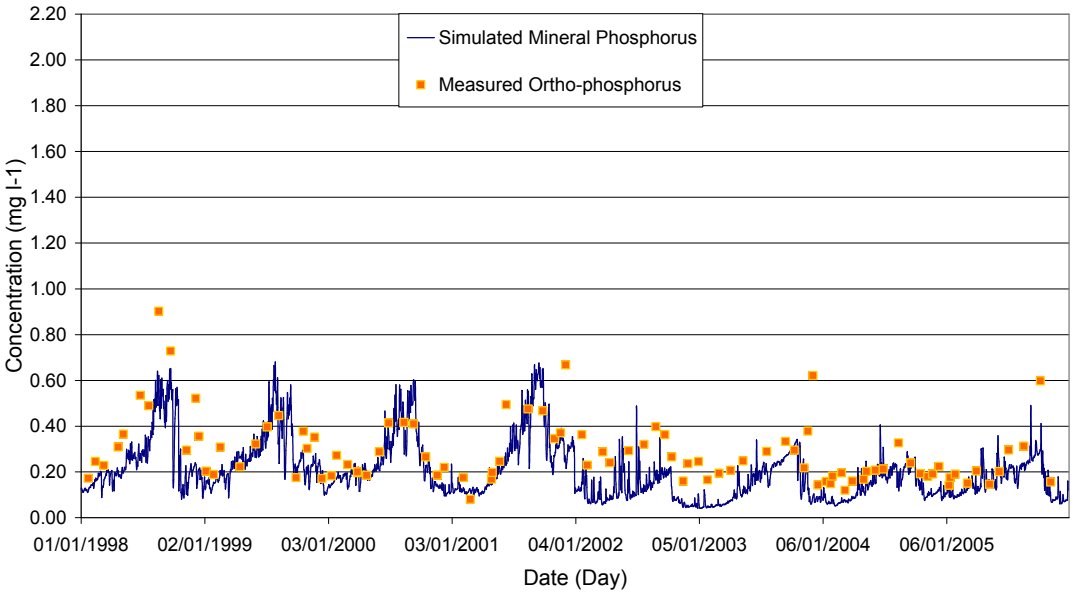
Appendix L6: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the Newenham Abbey sampling point during the validation period (1998 – 2005).



Appendix L7: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the Colyford sampling point during the validation period (1998 – 2005).

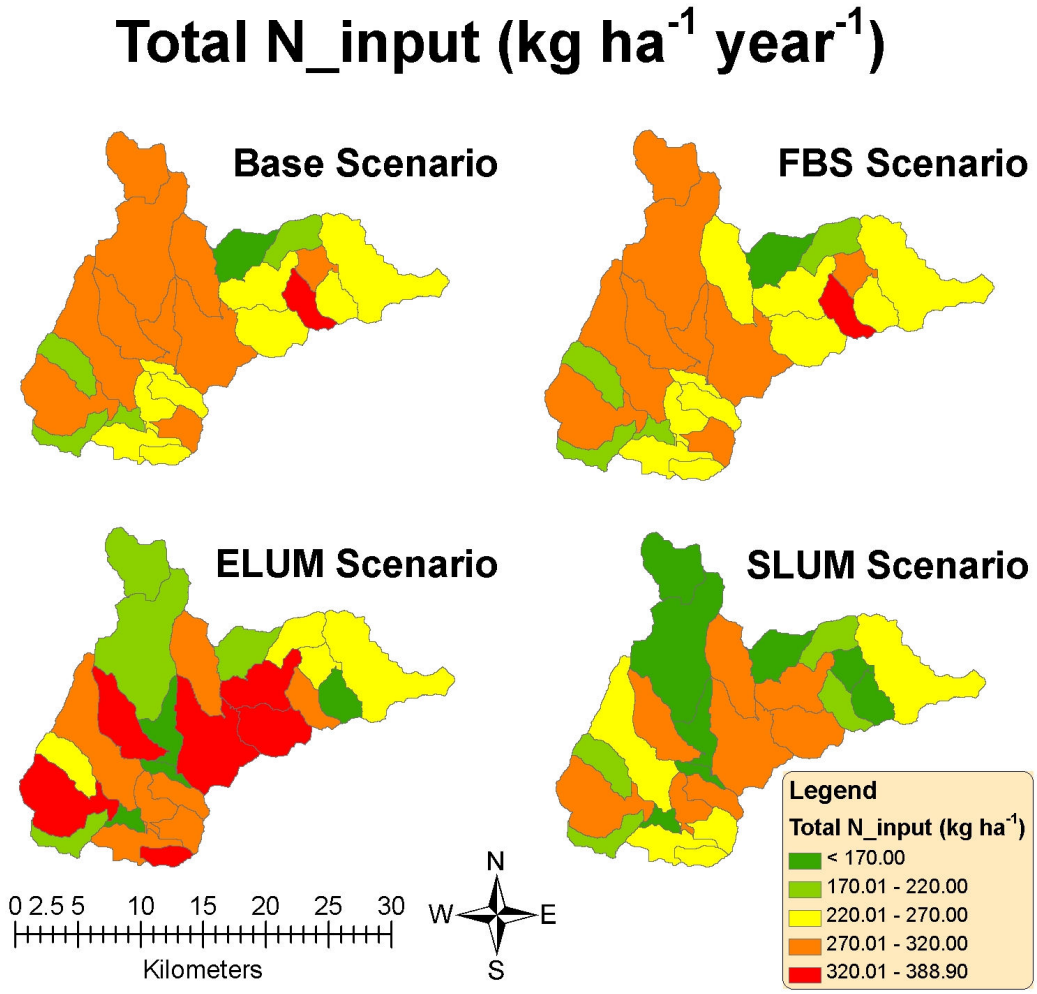


Appendix L8: Comparison between daily simulated mineral phosphorus (MINP) concentrations (SWAT) and measured ortho-phosphorus (PO_4^{3-}) concentrations (EA) for the Axe Bridge sampling point during the validation period (1998 – 2005).

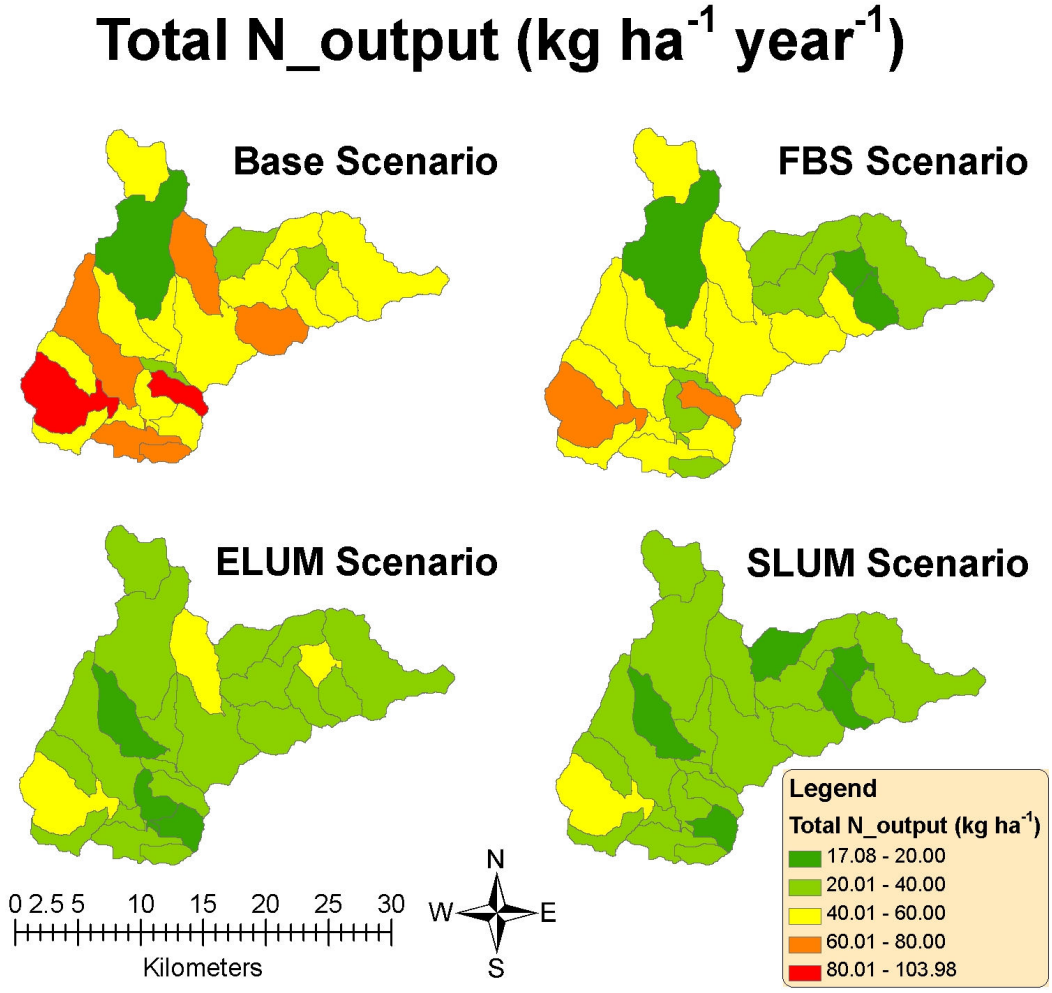


APPENDIX M

Appendix M1: Comparison of average annual total nitrogen inputs (kg ha^{-1} year^{-1}) at HRU level averaged to sub-catchment level for the baseline and land use management scenarios.

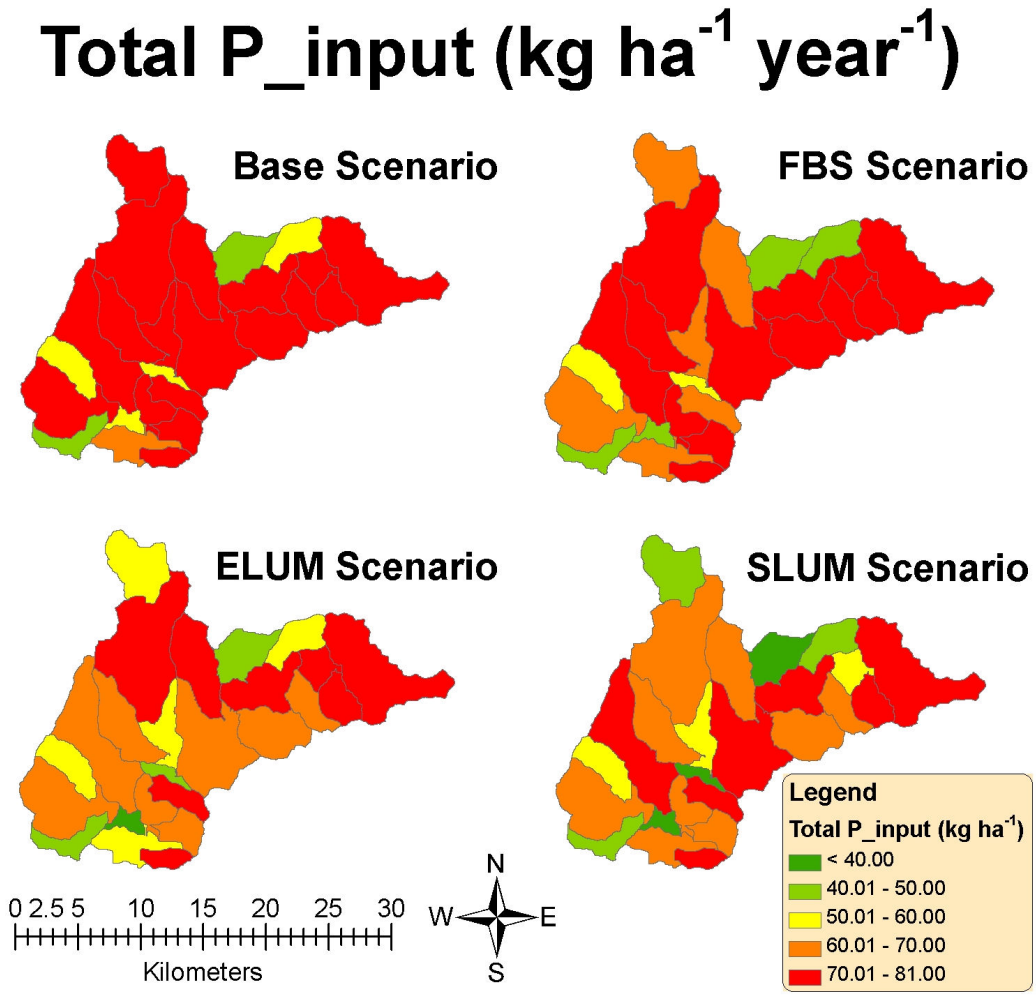
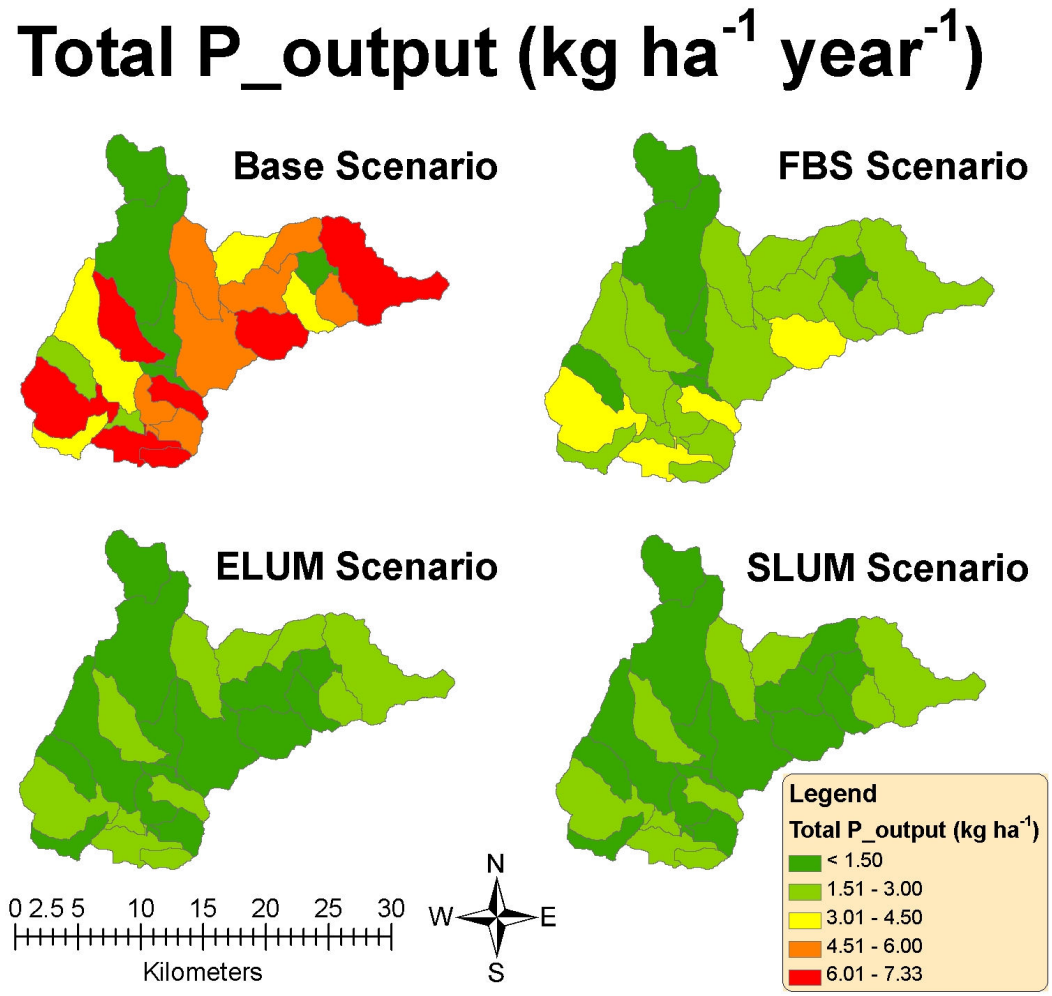


Appendix M2: Comparison of average annual total nitrogen outputs (kg ha^{-1} year^{-1}) at HRU level averaged to sub-catchment level for the baseline and land use management scenarios.



APPENDIX N

Appendix N1: Comparison of average annual total phosphorus inputs (kg ha^{-1} year^{-1}) at HRU level averaged to sub-catchment level for the baseline and land use management scenarios.



Appendix N2: Comparison of average annual total phosphorus outputs (kg ha^{-1} year^{-1}) at HRU level averaged to sub-catchment level for the baseline and land use management scenarios.