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Chapter · June 2016

DOI: 10.1002/9781119054450.ch6

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CHAPTER 6

Sustainable agricultural practices for water quality protection

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6.1 Introduction

The significant changes in conventional farming activities in the recent past are among the main reasons for environmental degradation, particularly from the effects on natural resources (Zalidis *et al.*, 2002; Santilocchi *et al.*, 2012; Pisante, 2013; Squire *et al.*, 2015). Erosion, salinization, compaction and reduction of organic matter represent the principal threats to soil health and indirectly to surface and groundwater quality. Water quality is indeed principally threatened by run-off, erosion and leaching.

Excessive irrigation or continuous rainfall leads to increase in infiltration rates, lowering water application rates due to enhanced water flow over inclined soil surfaces resulting in runoff (Jarvis, 2007; Gasso *et al.*, 2013). Consequently, water runoff carries the topsoil layer away with nutrients and contaminants that accumulate on the water surface, leading to eco-toxicity and eutrophication (Subbulakshmi *et al.*, 2009). One of the main reasons for runoff is soil compaction due to the decline of the water infiltration rate (Bai *et al.*, 2008); numerous research reports have shown this increment in water runoff when soils are compressed (Boulal *et al.*, 2011a,b; Silburn and Glanville, 2002; Silgram *et al.*, 2010). Traffic-induced soil compaction is identified as the practice, stimulated by sweeping of mobile farming units in which the soil particles are spatially reorganized increasing soil bulk density (Hamza and Anderson, 2005).

Soil erosion represents rapidly spreading phenomena, especially in environments characterized by dry seasons and heavy rainfalls, which adds to the dilapidation of farming terrain (García-Orenes *et al.*, 2009; Pisante *et al.*, 2013). Soil erosion has ecological effects on aquatic environments, which are externalized from the farming system (Stoate *et al.*, 2001). Degradation of arable terrain is aggravated by increased vulnerability of soil particles that get isolated and carried through erosive means, when any soil covering (living or dead mulch) is absent (Benites *et al.*, 2005; García-Orenes *et al.*, 2009).

Leaching into groundwater consists of transport through the soil pores of nutrients and agrochemicals suspended in water from precipitation or irrigation (Subbulakshmi *et al.*, 2009). This practice results in damage to actual water quality and is a source of eutrophication (Subbulakshmi *et al.*, 2009; Smith *et al.*, 2014). Some researchers report that it could be reduced by soil compaction owing to the diminution of the soil macro pore association (Boizard *et al.*, 2013); besides, it is also demonstrates that compaction decreases development of roots and their accessibility to nutrients, enhancing their losses (Gasso *et al.*, 2013) and, by limiting soil microorganism ability in decomposition, it favours the risk of agrochemical leaching (Chamen *et al.*, 2015).

Water Stress and Crop Plants: A Sustainable Approach, Volume 1, First Edition. Edited by Parvaiz Ahmad.

6.2 Principal water contaminants in agricultural systems

6.2.1 Sediments

Sediment is generally recognized as a main pollutant that necessitates enhanced management (Bilotta *et al.*, 2010; Collins *et al.*, 2012). A significant amount of soil determines changes in turbidity, light penetration, temperature and biological oxygen demand (BOD) in water systems, which can have severe harmful consequences on biodiversity (Watts *et al.*, 2003; Covich *et al.*, 2004; Greig *et al.*, 2005; Bilotta and Brazier, 2008). Applications between 80 and 400 mg L^{-1} are likely to directly influence the reproductive health and habitation of fish (Rickson, 2014) as well as indirectly by dropping the number of aquatic invertebrates in the food chain (Jones *et al.*, 2012; Bilotta *et al.*, 2012).

In addition, sediments also contribute to the transport and providence of numerous materials as well as nutrients, heavy metals, pesticides and added natural pollutants (Walling and Collins, 2008). The increased capacity of adsorption for adhered macro-elements in sediment was observed in soil-eroded particles like clay, slits and organic matter (Rickson, 2014). The delivery of such sediment to water is very dynamic because it depends on intricate interactions between sediment features and accessibility; erosion, transport and deposition methods (Parsons *et al.*, 2004; Govers, 2011); climatic changes (Reaney *et al.*, 2011); the source of innate and synthetic paths (Rickson, 2014); gradient extent and land utilization (Walling *et al.*, 2002); terrain supervision procedures, for example, soil management procedures (Govers, 2011) and the spatial allocation and concentration of the getting water streams (McHugh *et al.*, 2002; Watts *et al.*, 2003; Walling *et al.*, 2005; Gasso *et al.*, 2013).

6.2.2 Nutrients

6.2.2.1 Carbon

Water quality assessments provide evaluation of the following carbon fractions: total carbon (TC), inorganic carbon (IC), total organic carbon (TOC), particulate organic carbon (POC) and dissolved organic carbon (DOC) (Udeigwe *et al.*, 2011). POC is regarded as dynamic collection that manipulates nutrients discharge in farming regions (Kirkby *et al.*, 2013). The DOC portion has revealed to have an immense effect on soil organic activity (Kindler *et al.*, 2011), providing a principal resource of microbial substrate and, moreover, impinging on the carrying of heavy metals and natural contaminants from soil to surface waters (Udeigwe *et al.*, 2011).

The structure and quantity of carbon relinquished in runoff depend on size and constitution of the OM group, which is significantly increased by organization/farming and produces an inert and natural adjustment relevance (Udeigwe *et al.*, 2011; Aita *et al.*, 2012). On a large scale, it has revealed a stable increment in DOC application in lakes and streams (Driscoll *et al.*, 2003; Hejzlar *et al.*, 2003; Evans *et al.*, 2005; 2006).

6.2.2.2 Nitrogen

Agriculture has been pointed to as the main source of nitrate nitrogen (N) in surface water (Randall and Mulla, 2001). Nitrate N can be simply mislaid from the soil profile by leaching and, together with phosphorus, it is mainly responsible for water quality degradation. Surface runoff, leaching and favoured flow are the major procedures implicated in the allocation of N from soil to water. Inorganic fertilizers and animal manures are the main sources of N applied to agricultural soils (Randall and Mulla, 2001; Udeigwe *et al.*, 2011). The N forms, which are normally found to be present in water, comprise nitrate (NO_3^-), nitrite (NO_2^-), ammonia (NH_3) and organic N (Sawyer *et al.*, 2002). In newly contaminated water, ammonia and natural N are the chief N forms. Macrobiotic N is then progressively changed to NH_3–N that, in the presence of oxygen, might consequently be changed to NO_2^- and NO_3^- (Sawyer *et al.*, 2002).

The quantity and structure of N discharged from farming to the atmosphere is a rationale of the N supply, application rates, time and process as well as land use (Diaz *et al.*, 2009; Harmel *et al.*, 2009). Precipitation abruptly following external application of poultry manure can result in augmented runoff of whole Kjeldahl nitrogen (TKN, habitually utilized to characterize the amount of organic N and NH_3 available in a water system) and NH_4^+–N (Diaz *et al.*, 2009). Moreover, the TKN and NH_4^+–N application and fill in runoff is amplified by elevated compost rates, while manure assimilation significantly decreases runoff application and the stack of TKN and NH_4^+–N (Udeigwe *et al.*, 2011). The successful transfer of nitrate N to surrounding water takes place during subsurface drainage (tile lines) or base flow. Remarkably, diminutive nitrate N is mislaid from the terrain through surface runoff (Randall and Mulla, 2001). N concentration in water drainage depends on crops, N rates and application time (Qi *et al.*, 2011), that is, some studies indicated that N in water drainage in maise monocropping ranged around 14 mg N L^{-1}, while in perennial herbaceous crops (alfa alfa) was from 0.3 to 4 mg N L^{-1}.

6.2.2.3 Phosphorous

The excess of P in water causes problems associated with the accelerated eutrophication determining diminutive oxygen level, condensed aquatic species diversity, turbidity, disagreeable flavour and stench in public water stores (Greig *et al.*, 2012). The optimal organization of P for ecological and farming objectives is frequently incoherent since P losses from agricultural systems causes limited costs with respect to those from P inputs (Jørgensen *et al.*, 2013; Nordqvist *et al.*, 2014).

Soil P is originated beneath chemical and physical forms: soluble P, reactive P and stable P (Hansen *et al.*, 2002). The soluble pool is mainly reactive and plant accessible structure of soil P ($H_2PO_4^-$ e $HPO_4^=$) represents an extremely diminutive fraction of the overall P in soil, less than 1% (Hassan *et al.*, 2012). The other pools are strictly linked with the soil concrete segment and arise equally in natural and non-living forms (Sharpley and Rekolainen, 1997). The organic P comes from new natural matter, which quickly decays. The non-living P is found on soil exchange sites or in relatively soluble minerals.

Phosphorus coming into the soil can be carried in runoff water either in solution or linked with eroded soil particles. Conversely to the latter, P in solution is accessible for quick biological uptake, whereas bioavailability of P associated to soil can differ extensively depending on soil form, level of P saturation, particle size, organization account and redox potential (Sharpley and Rekolainen, 1997), ranging from 10 to 90%, with typical values around 20% (Eghball *et al.*, 2000; Uusitalo *et al.*, 2001; Hansen *et al.*, 2002).

The transfer of soil P to run-off water is a process occurring within a 1–5 cm perchè intensity of soil (Kleinman *et al.*, 2011) and is restricted by physical and chemical procedures, for example desorption, suspension and dispersion. The application P in solution run-off is reliably interrelated to the capacity and reactivity of P next to the surface of the earth (King *et al.*, 2014), different to the concentration of P associated to soil in run-off due to its dependence on the absorption of eroded soil particles.

In numerous watersheds, run-off and erosion is the main pathway of soil P loss (Benskin *et al.*, 2014) with insignificant subsurface P losses owing to elevated P fixing capacity of sub soils. Conversely, the movement of P in the course of the soil profile is imperative for those soils distinguished by an extremely little P sorption capability (minute quantity in clay Fe and Al oxides). Leaching of P is also exacerbated by heavy application of organic waste to sandy soils and acid innate soils. Sometimes, the continuing conduction of practical P matter to run-off or loss by restricted flow in leaching can be more imperative than the transfer of soil (Sharpley and Rekolainen, 1997; Greig *et al.*, 2012). The threat of steadfast conduction losses is the main a purpose of organization choice; for instance, at the instant of application, the supply of applied P and the application technique and tillage performance. Insertion or integration of applied P fertilizers or manure radically diminishes the hazard of through P loss in run-off.

6.2.3 Heavy metals

For a long time, agricultural soils have contained heavy metals, for example copper (Cu), zinc (Zn), lead (Pb), nickel (Ni), cadmium (Cd) and chromium (Cr). They are adsorbed by soil constituents and distributed by water runoff and leaching into rivers and lakes, resulting in pollution of imperative drinking water resources (Udeigwe *et al.*, 2011). Metals' mobility and bioavailability in the soil stratum are amended through organic matter (OM), clay minerals, pH, redox potential and Fe/Al oxides (McBride *et al.*, 2005).

Besides the atmospheric deposition, which is one of the major resources of significant metals flowing in arable terrain, livestock manures and sewage sludge characterize considerable resources in the majority of arable soils (Nicholson *et al.*, 2003). Agricultural fertilizers are also a way for the contaminations of arsenic (As), Cd, fluorine (F), Pb, mercury (Hg) (Jiao *et al.*, 2012) and sulfur (S) (Jarosiewicz and Tomaszewska, 2003). Increased concentrations of heavy metals, for instance Cu, Zn and Pb, have been linked with the utilization of agrochemicals in the majority of surface water of soil in China (Li *et al.*, 2014).

6.2.4 Agrochemicals

Agrochemicals penetrate the water surface mutually from point source contamination (e.g. Wesström *et al.*, 2014) and from disperse resources subsequent to their application to crops (Stoate *et al.*, 2001). The degree of pollution of agrochemicals in to water depends on their mobility, solubility and deprivation rate (Stoate *et al.*, 2001). On a farm scale, agrochemicals affect soil flora and fauna causing, as a consequence, both physical and chemical deterioration (Zalidis *et al.*, 2002; Stagnari, 2007). On a watershed scale, leaching, drainage and especially run-off of agrochemicals into the surface and ground water is a relevant issue, since they might be lethal to aquatic organisms and still potentially carcinogenic.

Modern agrochemicals degrade rapidly in soil but may persevere if they get into subsoil or groundwater owing to diminutive microbial activity, light deficiency and lower temperatures (Environment Agency, 2007). In dry-land crops, drainage can support the flow of agrochemicals to surface water, circumventing the soil stratum where their degradation mostly happens (Stoate *et al.*, 2001).

6.3 Best practices

6.3.1 Management of fertilization

Management of fertilization methods, executed to diminish fertilizer discharge into water, leads to the prime consequence that they are utilized at sites with an enhanced capability of damaging water quality.

Through adjusted fertilizer placement as well as best rate, timing of application, type of fertilizers and a tailored levelling, draining and contouring of farming terrains, economic and environmental benefits are obtained (Ruidisch *et al.*, 2013). Ridge tillage has the capability to diminish NO_3^- leaching, thanks to the possibility to place the fertilizer in the upper part of the ridges (Waddell and Weil, 2006). When the ridge tillage system is combined with plastic mulching, fertilizers are protected from infiltrating water and, accordingly, nutrient maintenance in the soil crest and nutrient use ability of crops are enhanced (Ruidisch *et al.*, 2013; Nguyen *et al.*, 2014). Besides, bare furrow sites are further susceptible to fertilizer leaching attributable to elevated infiltration rates originated through the surface runoff from the edge to the furrow (Leistra and Boesten, 2010).

In the case of P, application of fertilizers or manure without incorporation can favour steadfast P diffusion to exterior water, particularly when run-off follows abruptly; the amount of P loss increases with the application rate. The degree of P in run-off diminishes as the time linking application and run-off raises (Kleinman *et al.*, 2011). Besides, incorporation can diminish the absorption of dissolved P in run-off that instantaneously follows application up to fourfold (Chien *et al.*, 2011).

Applying the required plant amount of nutrients contributes to reducing water pollution by nutrients. It is acknowledged that a nutrient surplus, for instance nitrate, is liable to leaching from unnaturally drained fields (Dinnes *et al.*, 2002; Oquist *et al.*, 2007). Nitrification and mineralization of drop-applied nitrogen enhance the nitrate N content accessible for discharge in the subsequent spring (Dinnes *et al.*, 2002). P run-off can be condensed by circumventing redundant P application rates in susceptible regions, especially during high rainfall periods.

Fertilizer type also influences the quantity of nitrate N lost from the soil (Stagnari and Pisante, 2012). Inorganic fertilizers are accessible for crop uptake and also to leaching more quickly than natural ones: mineralization of organic N arises soon after in the emerging season (Thoma *et al.*, 2005).

Cropping practices are very effective in affecting nutrient losses from soils (Stagnari and Pisante, 2010). Annual row crops have been demonstrated to enhance thrashing of nitrate N and sediment-bound phosphorus than perennial species. This is accredited to better application of N fertilizers pooled with lesser evapotranspiration rates that effect in added drainage flow (Oquist *et al.*, 2007). Cover crops and legume-based rotations increase uptake and evapotranspiration thus decreasing soil nitrate-N that is minutely accessible for leaching (Dinnes *et al.*, 2002; Strock *et al.*, 2004; Culman *et al.*, 2013). Combining plastic mulch with fertilizer transfer constraints to edges and with split fertilizer spray can result in an 82% decrease in collective nitrate leaching (Ruidisch *et al.*, 2013).

A possible approach to advancing water quality by dropping fertilizer losses is the implementation of alternative agricultural activities. Substitute agricultural techniques comprise preservation, organization procedures, species biodiversity and/or activities that involve economical contribution of artificial fertilizer (Oquist *et al.*, 2007). Conservation tillage systems, by reducing sediment erosion, also limit the associated P and N losses whilst distinguished tillage strategies run off minute crop filtrate on the soil surface (Hansen *et al.*, 2002; Stagnari *et al.*, 2009). Moreover, to diminish the leaching threat of fertilizers and agrochemicals, meticulous farming is established as an expensive contrivance (Ruidisch *et al.*, 2013).

6.3.2 Erosion mitigation measures

Alleviation procedures (described as an 'on the ground structure or management practice by the land manager'; ADAS UK Ltd/Halcrow, 2008) valuable for soil erosion can be functional to manage dispersed contamination losses: indeed, the marked association among runoff and sediment with the transport of P, N, agrochemicals, pathogens and metals has been well demonstrated (Tyrrel and Quinton, 2003; Quinton and Catt, 2007; Bilotta and Brazier, 2008; Deasy *et al.*, 2009; Kerr *et al.*, 2011; Newell Price *et al.*, 2011).

Numerous kinds of alleviation procedures are identical in methodology (e.g. agricultural-planting-practice, strip rotary tillage, direct seeding, slot-planting, reduced-tillage-farming practice and direct drilling were classified as 'minimum cultivation systems') (Rickson, 2014). Several studies, especially conducted in UK, link to the efficiency of compact farming, of the use of buffer strips – riparian, of the condensed foraging force on soil thrashing and sediment assembly (Rickson, 2014). Several research reports demonstrate the association linking natural matter and soil erodibility (Munkholm *et al.*, 2002; Blair *et al.*, 2006a,b; Kadlec *et al.*, 2012). It emerges that the measures with highest effectiveness are represented by building of channels, mulching/tillage management practices, marshland characteristics, edge-of-field buffer strips, minimal cultivation systems (62%) and in-field grass buffer strips (Rickson, 2014).

6.3.3 Sustainable cropping systems

The interaction between sustainable cropping systems and water resources is very complex. Among the different sustainable cropping systems (i.e. organic agriculture, biodynamic, permaculture, precision farming, perennial crops and integrated agriculture) conservation agriculture (CA) has the higher potential for protecting water resources both in terms of quantity and quality (Stagnari *et al.*, 2010; Pisante *et al.*, 2012; Pisante and Stagnari, 2013).

CA plans to preserve, develop and formulate extra proficient utilization of natural resources during integrated administration of obtainable natural resources pooled with peripheral contributions. It adds up to ecological preservation in addition to improved and persistent agricultural productivity (Pisante *et al.*, 2007; Pisante and Stagnari, 2007). It could be classified as resource competent or resource efficient agriculture (Garcìa-Torres *et al.*, 2003). It is based on these principles and activities: preserving stable soil surroundings and advancing insignificant mechanical disturbance of soil during zero tillage systems; supporting vigorous, existing soil in the course of crop rotations, cover crops and the utilization of integrated pest management technologies and finally encouraging application of agrochemicals in equilibrium with crop necessities. The role of CA in reducing surface and ground water pollutants is clearly elucidated. In the US, CA has revealed to decrease runoff to about 15–89% and lowered suspended agrochemicals, nutrients and sediments contained by it (Lemke *et al.*, 2011). Cultivation influences the speed and quantity of rainfall infiltration and thus ground water revitalization, river flow rates and the requisition for irrigation (Boardman, 2013). In regions of scarce precipitation, CA assists in preserving water in the topsoil cover (Alam, 2014). Direct drilling pooled among stubble retention was shown to increase rain infiltration, leading to a reduced runoff compared to cultivated soil (Verburg *et al.*, 2012).

CA may have a depressing effect through evolution owing to the usage of herbicide for grass weed management (Ogle *et al.*, 2012) and because of escalating soil macropores that cause rapid movement of pesticides into the water course (Vryzas *et al.*, 2012). Conversely, the existence of earthworms can render elevated quantity of organic matter, which preserves agrochemicals and therefore assists in hindrance of pesticide release (Alletto *et al.*, 2012). Interesting research investigations demonstrated the diminished threat of agrochemical pollution in surface waters owing to implementation of CA in the USA. Direct drilling declined agrochemical runoff by 70–100% (Palm *et al.*, 2014) and leaching of isoproturon was lowered by 100% in a period of 3 years through CA (Jordan *et al.*, 2000).

6.3.4 Controlled traffic farming

Controlled traffic farming (CTF) is a managerial approach to reducing traffic-induced soil compaction (Raper, 2005; Gasso *et al.*, 2013). It is described as a 'crop production system in which the crop zone and the traffic-lanes are markedly and permanently alienated' (Gasso *et al.*, 2014) by utilizing in-field technology operational with steering facilities and self-routing structures (Raper, 2005; Bochtis and Vougioukas, 2008). The main positive aspect is the reduction of the trafficked region whilst contrasting with traditional procedures. The latter, during the cropping season can create a compressed surface of about 80–100% of entire fields in rigorous mechanical manipulation of soil and

30–60% in conservation tillage practices (Tullberg *et al.*, 2007; Gasso *et al.*, 2013; Shabtai *et al.*, 2014). Besides, CTF causes a trafficked portion of 10–20% of an entire field spot (Wang *et al.*, 2009; Tullberg, 2010) and it is demonstrated to reduce usage of fertilizers because of lesser nutrient disparagement (i.e. minute discharge, leaching and nutrient runoff) (Hao *et al.*, 2011) and recognition to augmentation of the root development and uptake of nutrients (Gasso *et al.*, 2013). Besides, crops underneath CTF can accomplish the equivalent yield with 20–30% lesser fertilization rates than beneath random traffic farming (RTF) (Gasso *et al.*, 2013; Chamen *et al.*, 2015). CTF can augment agrochemical investments (through 0.6–26%), once measured against RTF, attributable to a diminution of the pass to pass overlaps (Armengot *et al.*, 2015).

6.3.5 Buffer strips measures

Runoff can be effectively reduced with the use of vegetative buffer strips (VBS), preventing substances from entering the water stream and/or carrying away sediments, organic materials, nutrients and chemicals (USDA, 2000; Krutz *et al.*, 2005; Lacas *et al.*, 2005; Zanin *et al.*, 2009; Arora *et al.*, 2010). VBS are usually set up along creeks, streams, ponds or lakes to prevent pollution of their waters (Lacas *et al.*, 2005; Milan *et al.*, 2013). Their efficacy is generally expressed as a percentage reduction in plant protection product (PPPs) concentration in water; the literature reports VBS effectiveness generally above 50% (USDA, 2000; Rankins *et al.*, 2001). In addition to the riparian buffer strips, useful tools to mitigate diffusion of water pollution are in-field and edge field buffer strips, which are efficient in managing erosion by 54, 61 and 64% (Rickson, 2014). From 49 studies accomplished globally, 147 values of sediment elimination competence were obtained (Gumiere *et al.*, 2011), with a sediment removal efficiency ranging from 24 to 100%. Research work quantifying data on buffer strip performance indicate a high variability in sediment control effectiveness (Reichenberger *et al.*, 2007). Such high variability is to be found in a high number of factors involved (Table 6.1). The primary determinant of VBS efficacy is its design (Milan *et al.*, 2013) and the minimum width needed to achieve an acceptable level of effectiveness must be determined by slope steepness and correlated to its primary function (reducing sediment transport or increase infiltration). Secondary to design are numerous other factors affecting VBS effectiveness, such as surrounding cropland characteristics as well as the

Table 6.1 Factors affecting the erosion control effectiveness of vegetated buffer strips (slightly modified from Rickson, 2014).

Property		Influence	References
Nature of the pollutants	Particle size of sediments	Field data suggest that only sand and silt particles are deposited, while the fate of clay sized particles (with the highest concentrations of chemical contaminants) is less well defined	Kronvang *et al.*, 2005; Owens *et al.*, 2007
Properties of the buffer strip	Width	Trapping efficiency increases with width	Vinten, 2006
	Vegetation species	Broad, hairy leaves trap sediments more effectively than narrow, smooth leaves	Gill *et al.*, 2014
	Vegetation habit	Uniform swards are more efficient at trapping sediment	Wood *et al.*, 2006; Deeks *et al.*, 2012
	Age of buffer strip	Build-up of sediment and P may make buffer strips sources rather than sinks of diffuse pollution	Reisinger *et al.*, 2013
Slope gradient	Upslope and within buffer strip	On slopes, most sediment trapping occurs within the first 5 m	Vinten *et al.*, 2004
Soil type	Soil erodibility	Affects the degree of runoff volume and sediment in flow	Wood *et al.*, 2007; Posthumus *et al.*, 2013
Nature of the runoff event	Duration and magnitude of event	Redetachment of trapped sediment is more likely with longer, more intense events. Antecedent moisture conditions affect buffer strip effectiveness	Kronvang *et al.*, 2005
	Depth of flow relative to height of vegetation	Flow should not inundate the vegetation	Bear *et al.*, 2012; Stone *et al.*, 2013

environmental conditions (slope, microtopography, soil type, rainfall intensity, infiltration capacity, strip width and irrigation volume) (Müller *et al.*, 2004; Patakioutas and Albanis, 2004). Pesticide characteristics (solubility and persistence), as well as soil texture, organic content and crop and tillage management, also have a great influence (Patakioutas and Albanis, 2004; Arora *et al.*, 2010). Several species can be profitably seeded in a buffer strip; multi-species are normally preferable to those composed of a single one, because a combination of plant species generally results in stronger mitigation capacity (Milan *et al.*, 2013).

Is has also to be considered that VBS filtration activity can vary with the specific PPPs used, with the sediment amount carried by runoff water, water retention time in the VBS, soil infiltration rate and so on (Balderacchi *et al.*, 2013).

6.4 Conclusions and future prospects

Agricultural water quality has been recognized as a foremost ecological concern in developed countries and as a pertinent subject for plan scrutiny. The economic cost of agricultural water pollution is associated with removing pollutants (i.e. nutrients and pesticides from drinking water) to guarantee water provisions convene drinking standards, ecosystems health and commercial fishing, frivolous and cultural principles related to water resources. Increase in awareness among the cultivators implementing ecological arable farm administration procedures, including integrated pest management, fertilization management and adoption of mitigation measures effective for soil erosion control.

However, water policies addressing agricultural water pollution in the future should: (i) increase the municipal demands to decrease the vigour and ecological expenditures of water contamination from cultivation; (ii) use suitable blend of contrivances focused on concentration on cultivation water contamination concern, to guarantee the accomplishment of lucid farming, ecological and water strategy objectives, including advancement in arable farm organization and allied expertise, particularly biotechnologies and employing the geo-positional system (GPS); (iii) reinforce water plan improvements to present a vigorous dogmatic agenda; (iv) assist stakeholder contribution (i.e. farmers, industry and community groups) in the plan and deliverance of policy reactions for integrated water management, together with the progress of superlative activities and expertise to diminish water contamination; (v) evaluate the economic and ecological trade-offs (manure dispersal) and co-benefits (riparian buffers) among water contamination and further ecological strategies and (vi) inflate information (data) and knowledge (science) and facilitate public contact to this information to emphasize better policy making.

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