# BIRD INTERACTIONS IN HUMAN-DOMINATED LANDSCAPES

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## ABSTRACT

Birds being sensitive to anthropogenic mediated disturbances are thus regarded as one of the important indicators of environmental health. Monitoring environmental health needs understanding the bird community structure in relation to its habitat temporally and spatially. In today's scenario of climate change, deforestation and urbanization, future conservation of avian species can be ensured only if interaction of avian communities in different habitats is properly understood and habitat heterogeneity both at local and landscape level is managed. In general, land use changes from forest dominated landscapes to agricultural or urbanization and human dominated landscapes decreases bird species diversity decreases in human dominated landscapes. In the past decade though ecosystem degradation due to land use changes were increasingly recognized and agricultural and human dominated landscapes were identified as a viable alternate for biodiversity conservation. It is therefore, the present paper reviews the bird interaction in human-dominated landscapes to better understand the avian community structure, composition, distribution and its dynamics in these habitats.

**keywords:** Bird community; Bird-habitat interaction; Conservation; Management; Urbanization

## Introduction

Birds the feathered bipeds are omnipresent and are even intercontinental migratory visiting places for food and reproduction (Ali and Ripley, 1987). Their functional roles include pollination, seed dispersal, pest control, weed seed removal and nutrient cycling (Şekercioğlu, 2006, 2012; Whelan *et al*., 2008; Mäntylä *et al*., 2011; Maas *et al*., 2013; Ndang´ang´a *et al*., 2013) including their role as an important indicator of ecosystem health (Lawson *et al*., 1998; Gregory *et al*., 2003, 2008; Khan *et al*., 2013) including their causal factors (Şekercioğlu *et al*., 2004; Şekercioğlu, 2012). Increase in population and consequent urbanization or human domination, climate change and its effect like drought is disturbing and destroying the avian habitats which are threatening their existence with continuing decline of avian biodiversity (Rapoport, 1993; Parlange, 1998; Pickett *et al*., 2001; Hansen *et al*., 2005; Albright *et al*., 2010; Chen *et al*., 2011; González-Oreja, 2011; Şekercioğlu *et al*., 2012) with biotic homogenization i.e., replacement of diverse avifauna with few specialist species (McKinney and Lockwood, 2001; Crooks *et al*., 2004). Consequently, globally 1226 avian species were listed in the IUCN Red List as threatened including 88 Indian species (Anon., 2012).

Land use changes and management alter vegetation i.e. landscape structure (composition and configuration) or changes landscapes matrix deteriorating the habitats qualitatively and quantitatively in terms of food, water and cover for birds thus affecting their community structure and distribution (Western and Grimsdell, 1979; Fischer and Lindenmayer, 2006; Butler *et al*., 2010; Langgemach and Ryslavy, 2010; Pereira *et al*., 2010; Whittingham, 2011; de Baan *et al*., 2013; Defra, 2013; Schindler *et al*., 2013; Mimet *et al*., 2014). The type and intensity of land use and management at field scale or management of crops, at farm scale like changes in crop rotation and at the landscape scale or managing heterogeneity (Danhardt *et al*., 2010; Smith *et al*., 2010; Fischer *et al*., 2011, 2013; Guerrero *et al*., 2012; Morelli *et al*. 2013; Schindler *et al*., 2013) affect landscape matrices or composition and configuration which influence the bird community dynamics at agricultural landscapes (Uuemaa *et al*., 2013). Heavy pesticide and fertilizer use in today's intensive agriculture is another important cause of bird decline (Dhindsa and Saini, 1994; Pain *et al*., 2004). Holistic assessment on relationship between birds and habitats can be done through bird guild approach which give a clear understanding bird communities at a particular habitat (Brooks and Croonquist, 1990; Flade, 1994; O'Connell *et al*., 1998, 2000; Atauri and de Lucio, 2001; Bryce *et al*., 2002; Tscharntke *et al*., 2008; Karp *et al*., 2011; Marja *et al*., 2013; Morelli *et al*., 2013).

Studies of avian diversity in the human and agricultural dominated landscapes with species-specific roles and ecological services of diverse avifauna in various ecosystems are very few and particularly rare for institutional campuses (Dhindsa and Saini, 1994; Marzluff *et al*., 2001; Gopisundar and Kittur, 2013; Singh and Banyal, 2013; Sengupta *et al*., 2014). Little is known about the management of these ecosystems for conserving the avian life (Hennings and Daniel Edge, 2003). The prerequisite for management of ecosystems for avian conservation is proper understanding spatiotemporal dynamics of their community structure at a landscape level across geographical areas (Lee *et al*., 2004; Gopisundar and Kittur, 2013). Ensuring continuous avian conservation in present day scenario of habitat destruction and preventing their likely extinction is possible and viable through innovative ways of conserving them in the human or agricultural dominated landscapes but with proper understanding of bird habitat interaction and management of habitat at different landscape levels (Kheraa *et al*., 2009; Azhar *et al*., 2011; Rajpar and Zakaria, 2011; Bensizerara *et al*., 2013).

**Bird Assemblages and Community Structure**

Forest specialists constitute 53 % of the total tropical bird species, while 14 % of the tropical birds are agroforestry species and only 3 % agricultural specialists (Şekercioğlu, 2012). Bird assemblages and their community structure indicate structural and functional integrity and stability of an ecosystem (Bradford *et al*., 1998; Browder *et al*., 2002; Aich and Mukhopadhyay, 2008; Roy *et al*., 2011; Chatterjee *et al*., 2013; Hossain and Aditya, 2014) along with their economic and ecosystem services (Dhindsa and Saini, 1994; Borad *et al*., 2001; Şekercioğlu *et al*., 2004; Şekercioğlu 2006, 2012). This is because bird assemblages in a particular habitat depends on its available resources suitable for birds and is variable across geographical scale (Dhindsa and Saini, 1994; Borad *et al*., 2001; Basavarajappa, 2006; Kumar *et al*., 2006; Gopisundar, 2011; Gopisundar and Kittur, 2013; Kumar and Gupta, 2013). Globally in tropical agroforestry systems bird assemblages were disproportionately more frugivores and nectarivores and lesser insectivores than the forests (Tscharntke *et al*., 2008). Specialist birds in agricultural landscape have wider habitat and diet choice than the specialists in forest, thus with agricultural intensification diversity of insect feeding birds decreased, while avian pollinators and seed dispersers increased initially but proportionately decrease later (Tscharntke *et al*., 2008).

It was reported that occurrence of avian species in human dominated landscape is along a vegetation gradient ranging from truly urban environments to completely wooded areas (Rolando *et al*., 1997). Avian community varies across a range of habitats like agriculture farmlands, on trees, grasslands and other areas (Dhindsa *et al*., 1984, 1985; Braithwaite *et al*., 1989; Daniels *et al*., 1990; Gupta, 1994; Chakravarty, 1996; Chakravarty and Sandhu, 2002). Bird community structure was reported to be influenced by quality, structural diversity, disturbance level and food availability of the habitats at spatio-temporal scale both at local and landscape level indicating the dynamics of plant, insect and vertebrate population (Gregory *et al*., 2005; Firbank *et al*., 2008; Lindsay *et al*., 2013).

**Urban and suburban landscapes**

Globally urbanization is increasing which requires proper understanding of species distribution and abundance in the towns and cities across the globe (Chace and Walsh, 2006; Ortega-Álvarez and MacGregor-Fors, 2009, 2011). Globally in the cities 2041 species of birds (20 %) of the total bird species (10052) occurred represented by 144 families of the total 198 families (Aronson *et al*., 2014). Urbanization restricts the number and types of species to adopt and colonize the urban habitats (MacGregor-Fors and Schondube, 2012). Consequences of urbanization on wildlife are widely reported (Beissinger and Osborne, 1982; Koskimies, 1989; Bokotey, 1997; Fahrig, 1997; Rolando *et al*., 1997; Czech *et al*., 2000; Brooks *et al*., 2002; Kati *et al*., 2004; Scheifler *et al*., 2006; Furness and Greenwood, 2013).

Species are subjected with two options on urbanization either avoid or accept i.e., synurbization (Luniak, 2004; Biaduń, 2005). Drastic reduction in bird density was reported from 54 major cities of the world where only 8 % of native bird species as compared to non-urban region remained which was mainly due to anthropogenic driven factors like land cover and city age (Aronson *et al*., 2014). Diversity and composition of birds were changed by urbanization (Jokimäki *et al*., 2002; Blair and Johnson, 2008; Ciach, 2012; Møller *et al*., 2012). Urban bird communities were reported with high population densities and low species diversity (Baiten, 1972; Marchetti, 1976; Tatibouet, 1981; Taylor *et al*., 1987; Blair, 1996; Clergeau *et al*., 1998; Shochat *et al*., 2004) and species richness was reported to be affected by abundance and diversity of urban vegetation along with heterogeneity and disturbance in the habitats (Nuorteva, 1971; Lancaster and Rees, 1979; Petraitis *et al*., 1989; Dowd, 1992; Jokimäki and Suhonen, 1993; Natuhara and Imai, 1996; Clergeau *et al*., 1998). It was also reported that species richness increases with increased suburb or urban habitat age (Vale and Vale, 1976; Hohtola, 1978; Savard, 1978; Munyenyembe *et al*., 1989; Clergeau *et al*., 1998) while. no such effect was reported for city size (Luniak, 1990). Local habitat characteristics in large cities strongly influence the bird communities than its landscape setting (Huhtalo and Järvinen, 1977; Davis and Click, 1978; Luniak, 1990; Ortega-Álvarez and MacGregor-Fors, 2009; Stagoll *et al*., 2010; MacGregor-Fors and Ortega-Álvarez, 2011; Litteral and Wu, 2012).

Urban landscapes were reported with higher number of multiple brooder species those breed on urban structures, feed on seeds and residents with no territorial demarcation. This is exactly opposite for the birds residing at natural sites (Roy *et al*., 2012; Patra and Chakrabarti, 2014). Habitats of Suburban areas are in transitional state between heterogeneous natural habitats and homogeneous urban habitats (Blair and Johnson, 2008). Heterogeneity of urban habitats also was reported to support higher avian diversity due to availability of diverse resources (Pautasso, 2007). This indicates variation in functional roles, feeding habits and resource utilization pattern by the avian community of these urban habitats (Mahabal, 2005; Thakur *et al*., 2010). Urban parks with higher bird diversity than other urban areas are shelter and refuge for the birds (Rotenberry *et al*., 1979; Carbó-Ramírez and Zuria, 2010; Murgui, 2010; Strohbach *et al*., 2013; Tryjanowski *et al*., 2013) which sometimes are residual of native ecosystem are actually 'island' or 'oases and buffer for penetration for birds in urbanized landscapes (Senyk and Hornyak, 2003; Nagendra and Gopal, 2011).

Yard and gardens in residential landscapes contribute a major proportion of land cover in many cities across the globe (Loram *et al*., 2007, 2008; Clayton, 2007; Mathieu *et al*., 2007; Davies *et al*., 2009; Akinnifesi *et al*., 2010; Reyes-Paecke and Meza, 2012) provides a novel habitat for birds (Goddard *et al*., 2010, 2013; Lerman and Warren, 2011; Lerman *et al*., 2012$_{a, b}$, 2014; Fragkias *et al*., 2013). Private yards and gardens are 'oases' in urban or human dominated landscapes that provide variable structural features or otherwise scarce resources like trees, shrubs, grasses, rocks and water features to the birds (Chamberlain *et al*., 2004; Daniels and Kirkpatrick, 2006; Parsons *et al*., 2006; Bock *et al*., 2008; van Heezik *et al*., 2008; Burghardt *et al*., 2009; Evans *et al*., 2009; Ikin *et al*., 2013$_{a, b}$). Larger these gardens, diverse were the habitat features; more were attracted by the birds (French *et al*. 2005; Smith *et al*., 2005; Gaston *et al*., 2007; van Heezik *et al*., 2008; Davis *et al*., 2013, 2014; Kaoma and Shackleton, 2014; van Heezik and Adams, 2014).

In an industrial area of Panipat, Haryana, Dhadse *et al*. (2009) reported 63 bird species. Agricultural and horticultural landscape of Bengaluru region with ragi, rice, groundnut, sugarcane, castor, grapes and mulberry were documented with 38 eight bird species represented by 17 families and 26 genera with 22 resident insectivores, 12 resident migrants and four migrants (Rajashekara and Venkatesha, 2014$_a$). Bird abundance in the Bengaluru agricultural landscapes varied on the availability of variety of crops, nesting sites and perching trees.

A total of 125 avian species represented by 40 families were enlisted from the forest and urbanized habitats of Pauri District (Garhwal Himalaya) of Uttarakhand state, India (Naithani and Bhatt, 2012). Forests in Nainital district of Uttarakhand was reported with higher species richness (14.35 vs 8.69), higher species diversity (Shannon's index 4.00 vs 3.54), higher evenness (0.838 vs 0.811) and had more rare species (17 vs 5) than its urban habitat with abundance of 11 species higher in urban habitats (Bhatt and Joshi, 2011). In five distinct habitats of Udhampur region, Jammu and Kashmir 66 bird species represented by 11 orders and 27 families were reported with higher annual abundance of 904 birds and highest Simpson diversity index at urban areas (Singh *et al*., 2014). Terrestrial bird assemblages in a rural-urban gradient near the city of Amravati, on the Deccan Plateau, Central India was found with 89 bird species, with 67 species at rural landscapes and lowest of 47 at urban landscapes (Kale, 2014). Similarly, bird abundance decreased along the rural-urban gradient with stable species density throughout. In agricultural habitats with mixed cropping a total of 53 bird species were document which varied with crops on the field with bird damage (Bhale *et al*., 2012; Kale *et al*., 2012, 2013, 2014).

Urban parks of Puebla city metropolitan area in central Mexico were reported with 51 bird species reported were not strongly influenced by the habitat heterogeneity in terms of their community structure (González-Oreja *et al*., 2012). Riparian environments in Cai River, Rio Grande do Sul, Brazil was reported with 130 bird species with abundance, species composition and feeding guilds differed significantly among the riparian habitats. Generalist insectivorous species were more in the grassland and urban habitats, while leaf and trunk insectivorous and frugivorous were more in the woodlands (Brummelhaus *et al*., 2012). The bird species richness of 151 species was documented in Palmas urban area, Tocantins state, Brazil, however the study reported urbanization was decreasing the bird species affecting most of the trophic guilds and some families (Reis *et al*., 2012). Landscape variable like proportion of block area planned for residential use, area covered by unpaved roads and density of native trees were reported positively correlated with species richness, whereas density of commercial block, density of exotic trees and proportion of block area built in the Palmas urban area decreased the bird species richness.

Urban parks are important biodiversity hotspots in cities of Madrid, Spain and Oulu and Rovaniemi, Finland, however fragmentation had drastically reduced bird population in these cities (Fernández-Juricic, 2000$_{a,\ b}$; Fernández-Juricic and Jokimäki, 2001). Park size of about 10-35 ha was reported with higher species richness as compared to other habitats of the city. Linear vegetation features like street tree lines increased urban landscape connectivity and an alternate habitat during breeding season for food and nest. It was recommended to provide nest boxes and winter-feeding tables to increase bird diversity in the smaller parks of these cities. A total of 74 bird species found in the municipality of Örebro, Sweden was classified as woodpeckers, hole-nesters, forest birds and urban birds (Sandström *et al*., 2006). The city centre and residential area of the town was recorded with lower species richness than the greenway and periphery. Increasing rural-urban gradient decreased vegetation (amount and quality) as well as woodpeckers, hole-nesters and forest birds from town periphery to town centre but the trend of urban birds was reverse. Species richness of the bird groups except urban birds was found positively correlated with tree density.

Bird species richness and diversity of 63 bird species in three Swiss cities was negatively correlated with proportion of sealed area or buildings, while positive correlation was observed vegetation structures, i.e., mostly trees (Fontana *et al*., 2011). The bird groups reported from the Ukraine parks were breeding migrants, wintering migrants and residents. Revegetation in city of Brisbane, Australia increased the bird species richness by improving the connectivity between the remnant vegetation (Shanahan *et al*., 2011).

Bird species diversity, evenness of species abundances and numbers of species in urban habitats of Vancouver, B.C. was reported to increase with foliage height diversity and total vegetation (Lancaster and Rees, 1979). It was also observed that some man-made features improved a few bird niches. Bird species diversity and total bird density in urban areas of Vancouver did not decrease because of food provided by the residents. However, only a few cavity nesters, ground foragers and omnivores dominated the city. In three suburban conservation areas of the metropolitan

Vancouver 65 bird species were recorded, of which 39 species were urban adapters and six were exploiters (Mooney, 2011). Changes of resources due to urbanization cause differential response among the bird species as their composition changed from native species to invasive and exotic species in undisturbed areas of business district at Santa Clara County, California (Blair, 1996). In suburban neighbourhoods of Amherst, a university town at western Massachusetts, 64 bird species was observed during the breeding season (Degraaf and Wentworth, 1986). Bird communities responded to land use in the suburbanizing Twin Cities, Minnesota, USA due to changing landscape composition and habitat qualities depending upon the scale, type of habitat and component of the bird community (Chapman and Reich, 2007). After an initial increase with increasing urbanization, overall species richness, species evenness and Shannon diversity decreased significantly along rural-to-urban-gradients in three cities located at different eco-regions of USA (Blair and Johnson, 2008).

In urban areas of Sydney, Australia, Common Myna was found in 80 % of the sampled gardens indicating wide distribution of the species in the city followed by Rainbow Lorikeet (76 %), Pied Currawongs (64 %), Noisy Miners (59%) and Crimson Rosella were present in 45 % of the sampled gardens (Parsons *et al*., 2006). The native Australian species were recorded from less than 40 % of the sampled gardens (Willie Wagtail- 37 %; Eastern Yellow Robin- 7 % of the sampled gardens). In Adelaide, metropolitan area of South Australia 24 bird species was reported using street trees mainly the native trees (Young *et al*., 2007). Nectarivores were most abundant birds observed using the street trees. Tree species significantly influenced all the species of bird to use a tree including the dietary guild. Red gums were mainly used by the nectarivores, while insectivores used mainly plane trees. Use of trees by granivores varied with the season on availability of food. The birds were not influenced by the landscape features of the trees for traffic disturbance to the nectarivores.

In Canberra, 66 species of birds were recorded, of which 17 were native adapters, 20 were native avoiders, four were exotic adapters, one was an exotic avoider, 23 were native neutral species and one was an exotic neutral species (Ikin *et al*., 2013$_{a,\ b}$). Eucalyptus was planted in abundant (30 %) as street trees in suburbs. The reserves adjacent to these suburbs were reported with higher bird species richness and native adapter species richness. Bird species were positively correlated with habitat complexity rather than type of the street trees. Birds preferred native trees for foraging and thus Eucalyptus supported higher bird species richness as compared to exotic species and also aided adjacent reserves higher bird richness. Bird community in Wellington urban area of New Zealand constituted 35 species with House sparrow, Starling, Black backed Gull, Rock pigeon, Blackbird and Silvereye the most common and widely distributed species (Vinton, 2008). City landscape was found strongly correlated with species richness with highest richness green landscapes (n = 10, S = 15.9) and the least in wharf littoral and low-density commercial sites. Landscape diversity within an area was not found related to bird biodiversity, while bird abundance did not vary across the landscape of Wellington town.

In 54 wasteland sites distributed across entire urban area of Berlin, Germany 50 bird species were documented (Meffert and Dziock, 2013). The study found that the species with innovative behaviour was successful to thrive in densely populated city area as was indicated from higher adult survival rate of these species. Remnant natural and semi-natural areas in the Municipality of Rome, Italy were documented with 69 breeding bird species where species decreased with urbanization (Vignoli *et al*., 2013). Open habitat species decreased in abundance with increasing rural-urban gradient, forest species were neutral and generalist species increased with gradient. Predators and granivorous birds decreased with urbanization, while omnivorous birds increased in city scape of Rome. A total of 94 bird species were observed in Musanze city, northern Rwanda with no significant relationship of bird richness and relative abundance with city landscapes (Gatesire *et al*., 2014). Highest species diversity was observed in the residential neighbourhoods, institutional grounds and informal settlements. Urban parks in central urban area of Sendai, northern Japan were reported with 31 bird species during middle to late breeding season (Imai and Nakashizuka, 2010). The species richness was reported lowest in highly urbanized area with higher diversity index. Species richness was also strongly correlated with the presence of water-related environments in the Sendai City.

**Agricultural landscapes**

Crop diversity benefits to birds may vary spatially and can interact and confounded by heterogeneity of the landscape (Firbank *et al*., 2008; Henderson *et al*., 2009; Gabriel *et al*., 2010; Gottschalk *et al*., 2010; Tscharntke *et al*., 2012; Lindsay *et al*., 2013; Miguet *et al*., 2013; Ndang´ang´a *et al*., 2013; Palmu *et al*., 2014). Benefits of crop diversity may vary among different species of birds due to variation in resource, habitat and nesting preferences (Herzon and O'Hara, 2007; Filippi-Codaccioni *et al*., 2010; Gottschalk *et al*., 2010; Wretenberg *et al*., 2010; Hiron *et al*., 2013; Miguet *et al*., 2013; Ndang´ang´a *et al*., 2013; Chiron *et al*., 2014; Everaars *et al*., 2014; Sauerbrei *et al*., 2014). However, many farmland bird specialists prefer homogeneous open cropland landscapes with monocropping over diversified crops, while non-farmland birds prefer non-crop resources like forests, human habitation and wetland for their nesting and foraging needs over agricultural landscapes (Filippi-Codaccioni *et al*., 2010; Gabriel *et al*., 2010; Hiron *et al*., 2013). The non-farmland birds were thus reported to be benefitted from farm intensification which improve their resource or habitat availability (Filippi-Codaccioni *et al*., 2010). This is because the requirements of non-farm generalist's functional group like non-insectivores, vulnerable or endangered species and non-crop nesters are very specific habitat which may not fulfilled general crop diversification efforts.

Landscape heterogeneity and vegetation structure increases number of ecological niches within agroecosystems which enhances the richness and abundance of birds (Heikinnen *et al*., 2004; De La Montaña *et al*., 2006). Crop type and structural heterogeneity of an agroecosystem along with its management and landscape composition influences its preference by the birds (Verhulst *et al*., 2004; Taft and Haig, 2006; Bruggisser *et al*., 2010; Wretenberg *et al*., 2010; Karp *et al*., 2011). Crop diversity only influences bird diversity in simplified landscapes with inadequate non-crop resources (Wretenberg *et al*., 2010). Moreover, crop specific management like pesticide and fertilizer applications may affect bird diversity irrespective of resource availability (Guerrero *et al*., 2012; Jonsson *et al*., 2012; Palmu *et al*., 2014). Pesticide use intensification reduces the population of specialist farmland birds making them the most endangered group of birds (Gregory *et al*., 2005) while, crop-nesting bird population increases with landscape or non-crop management like local reduction of agricultural intensification (Guerrero *et al*., 2012). Organic farming benefits many bird species especially the granivores and insectivores (Christensen *et al*., 1996; Wilson *et al*., 1999; Freemark and Kirk, 2001; Beecher *et al*., 2002; Boatman *et al*., 2004; Piha *et al*., 2007) due to available food resources like increased weed and invertebrate abundance and richness with no pesticides (Wilson *et al*., 1999; Hyvönen *et al*., 2003; Mineau, 2005; Hyvönen, 2007).

Instead of crop diversity per se specific crop types like cereals, oil seed crops, etc. were reported attract farmland birds (Butler *et al*., 2010). Alternate to crop diversity, landscape heterogeneity was reported to mainly increase the dimensions of bird communities. Landscape heterogeneity is the function of non-crop and resource complementation or niche differentiation resources of habitats; presence of semi-natural habitat like scattered trees, field edges and hedge rows for foraging and nesting; accessibility to adjacent non-crop habitats for foraging due to smaller field sizes and lower proportions of cropland and minimum chemical inputs (Fahrig *et al*., 2011; Siriwardena *et al*., 2012; Josefsson *et al*., 2013; Lindsay *et al*., 2013). Diversified small farms creates habitat heterogeneity and negates biotic homogenization associated with large crop monocultures which supports bird diversity (Clavel *et al*., 2011; Karp *et al*., 2012).

Distance of agricultural landscapes to nearby forest creates a configuration effect that influences functionally useful avian and arthropod species like pollinators and predators (Klein *et al*., 2006; Perfecto *et al*., 2007). Closer the agricultural system to forest, more the pollinator and predator arthropod species richness which can not only augment crop yield (Kremen *et al*., 2002; Klein *et al*., 2003, 2006; Ricketts *et al*., 2004; Olschewski *et al*., 2006) but also led to attract the functional group of birds due to increased arthropods (Tscharntke *et al*., 2008). The diversity of forest generalists is more in agricultural areas closer to forests and with native forest cover than father away and without native forest cover (Klein *et al*., 2003, 2006; Sodhi *et al*., 2004, 2005; Soh *et al*., 2006; Laurance, 2007; Tscharntke *et al*., 2008). Closer to forest and native forest cover in agricultural areas attracts the forest species to agricultural landscapes (Harvey and Villalobos, 2007; Perfecto and Vandermeer, 2008; Gonthier *et al*., 2014).

Populations according to metapopulation theory are maintained by influxes to lower quality habitat patches with high quality matrix or land use types suitable for birds to feed and breed from source habitat (Vandermeer and Carvajal, 2001; Siebert, 2002; Dunford and Freemark, 2005; Bolwig *et al*., 2006; Philpott *et al*., 2008; Lira *et al*., 2012; Deikumah *et al*., 2013; Marcantonio *et al*., 2013; Villaseñor *et al*., 2014). Agricultural landscapes with at least some tree cover nearby are at a threshold level of hydrogenation and thus are able to attract forest generalist birds (Tscharntke *et al*., 2002, 2005). Generalist bird species thus are attracted to agricultural land-use systems within tropical mosaic landscapes which are connected to natural habitats by the presence of scattered trees, riparian or native vegetation and agroforests/homegardens (Elmqvist *et al*., 2003; Klein *et al*., 2003; Schroth *et al*., 2004; Bianchi *et al*., 2006; Tscharntke *et al*., 2008).

The potential of agroecosystems particularly the agroforests, homegardens, cash crop plantations, tree plantations or orchards as avian habitat is now increasingly recognised due to similar ecosystem services provided as that of forests. It is believed that about one third of all bird species is associated with agroecosystems but is a preferred habitat for a few only (Şekercioğlu *et al*., 2007; Şekercioğlu, 2012). Agricultural bird assemblages in the tropics are more generalists with multi-functional groups as compared to forests or tree plantations (Şekercioğlu, 2012). In an agricultural dominated landscape, the avian community is mainly dominated by a few granivorous, insectivorous and omnivorous species due to concentrated availability of food for birds like grains, seeds, fruits, green vegetation of the crop plants, grasses, weeds, insects, other invertebrates, and rodents (O'Connor and Shrubb, 1986; Toor *et al*., 1986; Dhindsa and Saini, 1994; Dhindsa *et al*., 1988; Chakravarty, 1996; Chakravarty and Sandhu, 2002; Asokan *et al*., 2009).

Avian species richness in an intensively cultivated area at Ludhiana was 68 species (Dhindsa *et al.*, 1988). In agricultural and other associated sub-habitats of Punjab, Malhi (2006) recorded 128 bird species. Gupta and Singh (2014) reported 79 bird species from agricultural landscape in Yamuna Nagar district of Haryana. Ardeidae was reported as the most diverse avian family in agricultural, sub-urban and wetland landscapes of India (Basavarajappa, 2006; Vijayan *et al*., 2006; Kumar, 2006; Gupta and Singh, 2014).

In shade-coffee and cardamom plantations and tropical rainforest fragments adjacent to Western Ghats Mountains of India 106 bird species were listed (Raman, 2006). Coffee plantations supported lesser rainforest bird species than the adjacent rainforest but the species richness found in the cardamom plantation was similar to the adjacent rainforest. Cardamom plantation was diversely intercropped with diverse native shade tree species. Plantations and fragments closer with each other offered connectivity to the rainforest birds; while those habitats with lesser connectivity or canopy cover attracted open-forest bird species. Habitats with woody plant variables, added more heterogeneity influenced bird community further. A study from agricultural landscape in Western Ghats, Maharashtra reported 97 bird species (Abdar, 2014).

The avian species assemblage of agricultural landscapes in Burdwan, West Bengal, India was reported with a species richness of 144 bird represented by 51 families and 19 orders with highest species richness in the order Passeriformes followed by Charadriidae and rest 17 orders. Residents dominated the list (61.15 %) followed by local migrants (31.65 %) and the least were migrants with 7.2 % of the total species reported (Hossain and Aditya, 2014).

Many workers reported that the diversity and species richness estimated in their study area were either higher (Soh *et al*., 2006; Lin *et al*., 2012; Sreekar *et al*., 2013; Subasinghe and Sumanapala, 2014); comparable (Ahmad and Yahya, 2010; Ahmed and Dey, 2014) from those earlier reported from tea gardens in India and elsewhere. The workers reported variation in the bird community across the studies could be due to different climatic conditions, elevation, area covered, shade trees, disturbance factors and proximity to primary forest of the respective study areas. The workers attributed high bird diversity in tea garden due to association of shade trees with tea bushes which provided heterogeneity in the habitat for fulfilling the multi-needs of the birds.

Small diverse farms, orchards and small woodlots in the Himalayas were reported to increase forest bird abundance during winter. In the traditional swidden cultivated landscapes at mountainous terrain, Xishuangbanna, Yunnan province of China, 148 species of birds were recorded in the Mengsong area and 107 species in the Jinuo area (Zhijun and Young, 2003) The bird diversity parameters were higher in Mengsong than Jinuo due to continuous various traditional land uses, while in Jinuo it was vanishing swidden agricultural system due to deforestation forest fragmentation. The differences in bird communities were due to both human disturbance and vegetation structure. Lower human disturbance attracted more farmland birds even if they were well-adapted to the disturbed agricultural environment.

The agricultural landscapes of south-western Poland including its field margins (Wuczyński *et al*. 2011) were each recorded with 50 breeding bird species. At the landscape scale, species composition differed between villages and the other environments, and villages were reported with more average bird abundance at landscape level, while at village level old homestead had more abundance.

A landscape level study by Piha *et al*. (2007) at an agriculturally dominated landscape of Pukkila in southern Finland found that total bird density, species richness, and diversity was non-significant with organic farming as potential effects of organic farming on higher food web levels are irrelevant in mosaic landscapes due to more availability of food for birds at nearby non-crop habitats (Bengtsson *et al*., 2005). Heterogeneity of habitat and crop diversity was found positively correlated with farmland birds in boreal cereal dominated agricultural landscapes of southern Finland

(Vepsäläinen *et al.*, 2005a, b; 2007; Vepsäläinen, 2007).

Small organic farms in temperate agricultural landscape of Sweden were reported with higher bird diversity of birds, pollinators and plants than large organic farms (Belfrage *et al.*, 2005). In forest fragments surrounded by farmland and natural forests of Uppasala, Central Sweden 50 bird species were observed where species-richness was positively influenced by both area and habitat heterogeneity (Berg, 1997). The worker reported that total bird density higher in fragments than in forest because of species foraging in the adjacent farmlands. Abundance of most the species was found influenced by the habitat quality variables (i.e., size, volume and diversity of tree species) and prominent was the presence of deciduous followed by tree diameter. This affect was found stronger in arable land than in the forest-dominated landscapes, while no such effects were found for field-nesting farmland bird species. Organic farming was reported to influence the field-nesting farmland birds only in the arable landscapes.

The bird communities in agriculture landscape with cereal crops was composed of 70 farmland and non-farmland bird's species (Chiron *et al.*, 2014). The study reported that with increased pesticide doses the proportion of habitat specialists particularly the herbivores decreased but the proportion of generalists increased. Total abundance and richness of birds increased with pesticides but no influence of insecticide or fungicide on birds were reported in this study. Increased pesticide dose indicated agricultural intensification of the study area which modified the bird communities by homogenizing species assemblages. In British farmlands the abundance of 12 species of common farmland birds declined, while 14 species increased between 1968 and 1995 due to changes in agricultural management (Siriwardena *et al.*, 1998). Data on birds occurring in farmland were related to the spatial organisation of farmed habitats in three different types. Species richness, abundance, and diversity of farmland bird communities in agricultural landscapes of Estonia, Latvia and Lithuania was reported positively correlated with farmland residual non-cropped elements, annual crop composition, grass fields and field types at landscape scale, of which stronger positive association was observed between farmland bird species richness and abundance and residual habitat richness and crops (Herzon and O'Hara, 2007). Landscape factors was reported for most of the variations in ground-nesting farmland bird individual and breeding pair densities visiting the largest cereal field available per farm mostly with winter wheat in Sweden, Poland, the Netherlands, Germany, Estonia and Spain (Guerrero *et al.*, 2012). This analysis found out that in general farmland bird densities were higher in simple agriculture dominated landscapes i.e., smaller fields with different crops but with reduction in cereal yield.

Forest reduction at the landscape scale caused drastic effect to communities of birds inhabiting the anthropogenic landscapes in the Brazilian Atlantic Forest, however, bird richness and abundance at the landscape scale was not affected by forest cover reduction sized when all species combined were considered. This was because different species responded differently to forest cover loss as there was a replacement of sensitive species with forest loss tolerant species along the gradient of forest cover change (Lindenmayer *et al*., 2005; Tscharntke *et al*., 2008; Lima *et al*., 2013; Bregman *et al*., 2014). The workers listed 184 bird species represented by 39 families in this Brazilian landscape with 60 % forest specialist species (103 species), while 56 species insectivores, and 34 species frugivores. Richness of forest specialist decreased sharply with decrease in forest cover but the generalists increased. Open habitats of central Brazilian Cerrado were observed with bird community of 110 species which changed significantly with the changes in vegetation along the gradient (Tubelis and Cavalcanti, 2001).

Reduction in population in these Costa Rican habitats were common in specialists, resident and insectivorous species. Moreover, the workers reported 49 % bird species preferred forest over coffee, 39 % preferred coffee over forest and 12 % preferred both. Coffee plantations of Costa Rica supported 185 bird species including some forest specialists also. Low intensity agroecosystem management like polyculture with high structural diversity was reported with resilient and stable as compared to high intensity agricultural management in Costa Rica (Karp *et al*., 2011). Vineyards in Hungary with diverse landscape elements like shrubs supported higher bird richness (Verhulst *et al*., 2004) but in Italian vineyards birds were more attracted towards its matrix than on the vineyard itself (Laiolo, 2005). Jones (2014) reported 44 bird species in the three distinct agricultural landscapes in Guadalupe, Panama with diversity differing significantly among the three landscapes. Of these reported species, 5, 3, and 6 species were specialists to forest corridor, pasture and forest edge and the rest were generalists. Among the three habitats, forest edge was estimated with highest species diversity and evenness followed by the forest corridor and pasture site.

Philpott and Bichier (2012) observed 113 bird species in coffee agroecosystems of Chiapas, Mexico. The workers found similar cumulative bird richness in both cut and uncut coffee plantations but abundance and mean richness was 3-6 times higher in uncut areas. It was reported that more the depth and cover of the coffee canopy more was the diversity and abundance of birds in them. Raptor diversity at agricultural landscape in northern-central Mexico of the Highland plateau of San Luis Potosí and Zacatecas was reported with 14 diurnal raptor species with no significant variation among landscape types. Bird community analysis at cattle grazing lands, crop fields, urban areas and riparian habitats in highly human modified landscape of north-western Colombia found 57 species (Domínguez-López and Ortega-Álvarez, 2014). Grazing lands were reported with diverse bird communities due to presence of tall trees with abundant shrubs and closeness to riparian habitats. The authors concluded that in the human-dominated landscapes of Columbia presence of riparian habitat was crucial to support diverse bird communities. Only a few species were dominant in the crop fields and urban areas which accommodated these disturbed sites.

Greater the species richness of shade trees in cacao agroforestry of Indonesia, higher was the species richness of birds in it, while herbaceous vegetation was neutral on birds (Clough *et al*., 2009). Overall, 56 bird species was recorded from these cacao agroforestry plantations. It was also reported that frugivores and nectarivores richness decreased as the distance of cacao agroforestry increased from the forest, while richness of granivorous birds increased with distance from the forest. Density of taller trees attracted all the functional groups except the seed eaters. Moreover, it was also observed that forest specialists were positively influenced to forest edge proximity. Associated shade trees in cacao agroforestry attracted birds independent of distance to forest. In oil palm plantations of Jambi province, Sumatra, Indonesia 33 bird species recorded (Teuscher *et al*., 2015) were reported highly extremely impoverished compared to natural forests (Peh *et al*., 2006; Teuscher *et al*., 2015). It was reported that though bird diversity and abundance was related positively and non-linearly to numbers of remnant or planted trees but negatively with oil palm yield.

Bird species diversity and richness of foraging guilds except insectivores increased as heterogeneity of vegetation cover types increased in the highland agricultural landscape of Nyandarua in Kenya (Ndang´ang´a, 2013). Crop diversity was also found positively correlated with bird species richness but cereal cover decreased species richness, overall abundance and granivorous abundance in this Kenyan highland agricultural landscape. Kenyan highland agroecosystems were also dominated by ground foraging birds regulating weed population, while aerial foraging insectivorous birds were dominantly feeding on invertebrate controlling insect pest population (Ndang´ang´a *et al*., 2013). Plantation forest had the lowest relative richness was lowest in the plantation forest. Forest specialists also were found in the remnant natural forest of Mount Kenya. Bird species richness was more in shaded coffee (or agroforestry) plantations than the sole or sun coffee in India, Neotropics and Africa (Wunderle and Latta, 1996; Greenberg *et al*., 1997a, b, 2000; Calvo and Blake, 1998; González, 1999; Moguel and Toledo, 1999; Petit and Petit, 2003; Donald, 2004; Mas and Dietsch, 2004; Perfecto *et al*., 2004; Komar, 2006; Raman, 2006; Gordon *et al*., 2007; Anand *et al*., 2008; Gove *et al*., 2008; Philpott *et al*., 2008; Rao, 2011). Higher bird abundance and richness especially the understory insectivores and omnivores on these coffee farms were also related to pest control and higher coffee yields (Greenberg *et al*., 2000; Perfecto *et al*., 2004; Kellermann *et al*., 2008; Van Bael *et al*., 2008; Philpott *et al*., 2009; Johnson *et al*., 2010; Karp *et al*., 2013; Railsback and Johnson, 2014). In East Africa, bird species richness was reported higher in mixed agriculture than forests (Mulwa *et al*., 2012) because East African agricultural landscapes were structurally complex due to adjoining intact forest and higher native tree density, crop diversity and hedge volume (Naidoo, 2004; Gove *et al*., 2008; Otieno *et al*., 2011; Mulwa *et al*., 2012). Therefore, in East Africa birds were reported to increase coffee yield by 9 % (Classen *et al.*, 2014).

Sun and shade coffee farms were recorded with 77 species, including 24 species of omnivores, 19 species granivores, 35 species insectivores, 3 species nectarivores and two species frugivores (Smith *et al.*, 2015). In these Kenyan coffee farms effect of local landscape variations was non-significant on bird abundance in the presence of arthropod diversity particularly on the omnivores and insectivores along with effect of scale fragments or landscape composition. The authors cautioned the coffee growers and managers in Kenya to balance management action for coffee insect pests and protection of insectivorous birds. Intercropping systems in Kenya supported higher bird richness and abundance than high intensity sole sugarcane (Mulwa *et al.*, 2012).

**Institutional campus**

Institutional campuses were reported to be a preferred habitat for birds and the diversity may increase if its landscape and vegetation are properly managed. Avian species richness in Kurukshetra University was 92 represented by 37 families with 71 resident species (Gupta *et al.*, 2009). Sixty-two avian species were reported to be associated with the orchards and its surrounding windbreaks (Chakravarty, 1996), while 21 species were associated with plant nursery and cropping areas near residences (Islash, 2010) in the premises of Punjab Agricultural University, Ludhiana. Indian Institute of Forest Management (IIFM), Bhopal was reported to host 106 bird species represented by 52 families, of which 27 species were winter visitors (Aggarwal *et al.*, 2015). Density of birds estimated in the campus was 32.7 birds/ha.

**Wetland and water bodies**

Wetlands the “biological supermarkets” are complex and productive ecosystems rich in biodiversity and an important bird habitat (Maltby, 1986; Weller, 1999; Mitsch and Gosselink, 2000; Stewart, 2001; Prasad *et al.*, 2002; Unni, 2002; Urfi *et al.*, 2005; Datta, 2011). The Ahiran Lake in Murshidabad district, West Bengal was recorded with 30 species of water birds representing 29 genera and 12 families, of which 16 were migrants and 14 residents (Mistry and Mukherjee, 2015). The wetlands of Birbhum district of West Bengal were reported with 25 bird species represented by nine families and 20 genera (Gupta and Palit, 2014). Among these listed bird species 60 %) were common, 32 % uncommon and 8 % were less common in the wetlands while, 40 % of the total species listed was reported with unknown status, 36 % decreasing and 12 % each with stable and increasing status.

In Bengaluru city, of 42 water bird species, diversity, evenness and richness varied among different lakes. Anthropogenic disturbances had negative and water depth had positive correlation with the bird density in the urban lakes of Bengaluru city (Rajashekara and Venkatesha, 2014$_b$). In this study, the species richness and community structure of irrigation ponds were characterized on the local and landscape scales. Open-canopy ponds were more heterogeneous than overgrown ponds and thus were used by diversified birds. However, overgrown territorialised ponds were used by some woodland bird species.

**Bird Interaction with Habitat**

Interaction of bird species with habitats along environmental gradients was mainly reported from temperate countries (Bond, 1957; Nuorteva, 1971; Cody, 1974; Able and Noon, 1976; Lancaster and Rees, 1979; Jokimäki and Suhonen, 1993; Blair, 1996, 2001, 2004; Mckinney and Lockwood, 2001; Adamik *et al*., 2003) and hardly any such studies in India was reported (Bhatt and Joshi, 2011; Naithani and Bhatt, 2012). The bird abundance and distribution are a direct function of ecosystem structure and composition (Cody, 1974, 1978; Ripley, 1978; Weins, 1989; Van Strien, 1997; Dorazio *et al*., 2015). Avian diversity varies among habitats as its diversity is directly linked with plant diversity and not to its abundance (Das, 2008; Singh *et al*., 2014).

However, bird population size is unaffected by tree diversity (Das, 2008). Forests or any other heterogeneous vegetation supports large number of bird species because of its diverse plant population (Naithani and Bhatt, 2012; Singh *et al.,* 2013$_{a,\ b}$). Diverse plant population increases the heterogeneity or complexity of habitat which increases the availability of food for the birds which increases their diversity and population. For example, an agricultural landscape with scattered trees and hedgerows increases bird diversity (Parish *et al*., 1994; Sierro *et al*., 1994; Hinsley and Bellamy, 2000; Whittingham *et al*., 2001, 2009; Berg, 2002; Padoa-Schioppa *et al*., 2006; Martin *et al*., 2012; Hiron *et al*., 2013). Enhancing landscape connectivity through revegetation was reported to have greater influence than patch area on bird species richness within urban revegetation and combination of both strongly influence bird abundance (Beissinger and Osborne, 1982; Shanahan *et al*., 2011). This is because connectivity increases the number of habitat patches (i.e., effective vegetation area) where colonists can survive.

Small wetlands particularly ponds were also reported crucial for birds in any landscape as they aid to increase habitat heterogeneity at the landscape level (Williams *et al*., 2004, 2010; Declerck *et al*., 2006; Davies *et al*., 2008; Ruggiero *et al*., 2008; Céréghino *et al*., 2008, 2014; Lemmens *et al*., 2013). Diverse population aquatic macro-invertebrates thrive in farmland ponds which were reported as preferred food for nesting and fledging birds including wintering water birds (Newton, 1998; Baxter *et al*., 2005; Richardson *et al*., 2010; Schummer *et al*., 2012; Matuszak *et al*., 2014; Stenroth *et al*., 2015). Larger the pond, more the bird species it attracts (Froneman *et al*., 2001; Sebastián-González *et al*., 2010) because of abundant and spatially heterogeneous macrophyte communities attracting birds for food, nesting material, habitat and refuge (Cody, 1981, 1985; McKinstry and Anderson, 2002; McAbendroth *et al*., 2005; Santoul *et al*., 2009; Sebastián-González *et al*., 2010; Thomaz and da Cunha, 2010; Florencio *et al*., 2014). Ponds surrounded by grasses and other plants also attract granivores to forage on the seeds (McCracken and Tallowin, 2004). Avian community structure at different habitats vary due to variation in availability of food to the birds, nesting sites, change of climatic conditions and consequent emigration and immigration

(Singh *et al.,* 2013$_a$; Pearce-Higgins *et al*., 2015). The availability of food resources in a habitat by birds determines its trophic structure in the community (Karr *et al*., 1990). Selection of diet composition by different was reported relevant to the tests of niche or guild concept (Wiens, 1989). Bird communities were also reported to be influenced by climate change as there were significant altered species' interactions to climate change impacts (Pearce-Higgins *et al*., 2015).

Land use change to from natural forests to agricultural urban lands for development (Hansen *et al*., 2013; Narayana *et al*., 2013) resulted into homogenous, dense, artificial environments has decreased the bird species diversity and bird species richness (Emlen, 1974; Aldrich and Coffin, 1980; Beissinger and Osborne, 1982; Pimm *et al*., 2006; Naithani and Bhatt, 2012; Batáry *et al*., 2014; Katayama *et al*., 2015) but can increase the abundance of some bird species (Huhtalo and Jarvinen, 1977; Clergeau *et al*., 1998; Naithani and Bhatt, 2012). Urban development like roads and highways had significantly decreased bird assemblages (Delgado García *et al*., 2007; Palomino and Carrascal, 2007; Fahrig and Rytwinski, 2009; Griffith *et al*., 2010) as compared to natural or rural areas (Matson, 1990; Clergeau *et al*., 1998; Vandermeer *et al*., 1998; Crooks *et al*., 2004; Laube *et al*., 2008; Batáry *et al*., 2014). In urban areas noise pollution was reported one of the prominent factors reducing bird diversity and abundance due to disturbances during mating, predator evasion and other activities (Reijnen and Foppen, 1994).

Roads, railways and several associated constructions were mainly related with loss of biodiversity but were also reported with some positive effects on certain bird species or communities (Morelli *et al*., 2014). Firstly, the roads are foraging habitat with less predation pressure and their warm and its warm surface help birds to conserve metabolic energy. Secondly, street lights prolong diurnal activity. Thirdly, power transmission lines and fences are used as perches during predation and lastly, birds use bridges, pylons, street trees, bases of power line pylons and green roofs for breeding and cover from predators. Significant differences were observed between native and non-native street trees with respect to higher prevalence of birds and nests with native trees supporting more species in higher densities especially reducing the negative effects of urbanization (White *et al*., 2005; Fernández-Juricic, 2000$_{a,\ b}$). The size of the trees and its proximity to birds was reported determinant for species variation in them. Positive relationship was observed between street tree species richness and bird species richness at street scale.

Forests were reported with higher bird diversity than urban habitats at all altitudes (Weins, 1989; Cody, 2001; Naithani and Bhatt, 2012). However, composition and structure of bird community may vary with elevation of a particular habitat due to variation in physical environment and availability of resource required for breeding and foraging (Able and Noon, 1976; Cody, 1981). Moreover, variation in species richness across an altitudinal gradient was also attributed to combined effect of many local and regional factors (Rahbek, 1997; Lomolino, 2001). Seasonal variation of bird species diversity and richness along altitudinal zones was also reported due to altitudinal migration of birds especially during winter from higher to lower elevation to avoid harsh weather conditions and deficiency of resources (Naithani and Bhatt, 2012). Seasonal variation in temperature and rainfall/precipitation causes inter- and intra-habitat migrations (Vázquez-García and Givnish, 1998; Norris and Marra, 2007).

Habitat specialists were reported the most vulnerable and thus less abundant, while generalists were successful in any or changed and disturbed habitat like human dominated landscapes due to their lower body mass and wider diet preferences (Głowaciński, 1990; Skórka *et al*., 2006). Ground foraging bird species prefer habitat with open canopy or bare ground having available weed seeds or other foods (Moorcroft *et al*., 2002; Schaub *et al*., 2010). Insectivorous bird species were reported dominant in forests and agricultural including tea and other plantations or human-modified landscapes (Chettri *et al*., 2005; Ahmad and Yahya, 2010; Chettri, 2010; Bhatt and Joshi, 2011; Singh *et al.,* 2013$_a$; Sreekar *et al*., 2013; Ahmed and Dey, 2014; Kottawa-Arachchi and Gamage, 2015). Bird diversity and its abundance varies with different agroecosystems due to variation in crop composition, farming intensity and availability of food, roosting and nesting sites, predation pressure and human disturbance (Cunningham *et al*. 2013). Seasonal abiotic variations which affect shelter and food availability influence avian communities in temperate agroecosystems (Gutiérrez *et al*., 2010; Cox and Underwood, 2011; Kelt *et al*., 2012). The seasonal effect increases diversity and abundance in winter at habitats with mild weather conditions that cause horizontal or vertical migrations (Cody, 1970; Kelt *et al*., 2012). Agricultural landscapes attract birds during winter as it provides alternate feeding grounds to the birds (Figueroa and Corales, 2005; González-Acuña *et al*., 2013).

Bird diversity greatly varies in agricultural landscapes due to its different land use i.e., configuration or patchiness of the crop land however, this relationship was reported very dynamic due to varying functionality and structural heterogeneity of agricultural landscapes i.e., more the heterogeneity greater is the bird diversity (Dunning *et al*., 1992; Fahrig *et al*., 2011; Mitchell *et al*., 2014$_{a,\ b}$). Few species were reported to use agricultural habitats but may not depend on it (Pimentel *et al*., 1992; Şekercioğlu *et al*., 2007; Sutcliffe *et al*., 2015). It was reported that productive landscapes like that of agriculture or plantation based were dominated by forest generalists while forests by the specialists (Tscharntke *et al*., 2005; Rand *et al*., 2006; Mastrangelo and Gavin, 2014; Carrara *et al*., 2015).

Farming systems with mosaics of field margins i.e., with linear features of high vegetation like tree lines and hedge rows are rich with diverse bird communities (Sanderson *et al*., 2009; Whittingham *et al*., 2009; Wuczński *et al*., 2011). Less intensive management of agricultural and other land use will improve the habitat matrix (Dietsch, 2005; Kennedy *et al*., 2010; Ruiz-Guerra *et al*., 2012; Deikumah *et al*., 2013, 2014; Sanderson *et al*., 2013) which will reduce threshold effects by increasing heterogeneity with more connectivity and fragmentation (Tscharntke *et al*., 2002, 2005; Devictor and Jiguet, 2007), thus access to food and breeding site (Antongiovanni and Metzger, 2005). Less intensive agricultural practices with hedge rows and agroforestry can create the edge effect (Perfecto *et al*., 2007). However, patch size of these vegetation mosaics (i.e., individual farm management unit) along with the length and density of linear vegetation in the landscape will affect matrix quality (Fischer and Lindenmayer, 2005; Scozzafava and De Sanctis, 2006; Billeter *et al*., 2008). Choice of species for a large plantation with a single owner may be uniform, whereas the species choice may be different in a similar sized landscape of small farms along with inter-boundary vegetation of the farms. Because of this management heterogeneity in different smaller farm units of a landscape create mosaics of quality matrix supporting diverse bird species (Perfecto *et al*., 2004, 2007).

Maintaining even 5 % native canopy cover on an agricultural landscape attracts 100 % of forest edge bird species (Peh *et al*., 2006). For example, scattered fruit trees in agricultural areas are important food resources for frugivorous birds even from nearby forests (Luck and Daily, 2003; Şekercioğlu *et al*., 2007). Moreover, closer proximity of agricultural habitats to forests strongly influences its functional diversity of agroecosystems (Tscharntke *et al*., 2008). Agricultural landscapes with diverse plant types support diverse bird population due to abundant food and microhabitat segregation for the birds (Hossain and Aditya, 2014). However, traditional agroforestry and tea plantations especially those closer to forests also support diverse avian species including the specialists also (Greenberg, 1981; Pimentel *et al*., 1992) as a corridor between the forests (Sreekar *et al*., 2013; Kottawa-Arachchi and Gamage, 2015). Native vegetation (riparian vegetation, forest fragments and forest) and diversified farming system (agroforestry and crop rotation) support diverse bird community (Henderson *et al*., 2009; Kremen *et al*., 2012; M'Gonigle *et al*., 2015).

The maximum numbers of birds in an urbanized landscape of Rajasthan were omnivorous followed by insectivorous and carnivorous (Joshi and Bhatnagar, 2015). In Indian Institute of Forest Management (IIFM), Bhopal bird species were classified into eight feeding guilds: carnivore, ground insectivore, sallying insectivore, canopy and bark insectivore, nectar insectivore, general insectivore, frugivore and water birds (Aggarwal *et al*., 2015). Diverse populations of birds along altitudinal gradient and habitat types were also reported from Garhwal Himalayas where distribution and diversity indices increased with increasing elevation in both forest and urbanized habitats (Naithani and Bhatt, 2012). Moreover, this Garhwal study also reported lesser overall bird species richness and bird species diversity in urban habits than in forests but found higher relative abundance of seven bird species in urban habitats than the forest habitats. Bird species dominance in this Garhwal study was reported to vary in forest habitats along altitudinal zones but in urban habitats

House Sparrow (*Passer domesticus*) was found dominant species throughout the altitudinal zones.

Most of the bird species (51.85 %) reported in a study from Burdwan, West Bengal was reported to use agricultural fields as their habitat followed by aquatic systems (29.20 %) and the lease was human habitat with 18.98 % of the bird species recorded (Hossain and Aditya, 2014). Similarly, Ahmad and Yahya (2010) reported 130 bird species of birds in six different habitat types of Kurseong Hill, of which 48 were found in the productive tea plantations. The size of the tree gardens with diverse tree species and abundant arthropods supported the bird community to interact for their needs satisfactorily (Hazarika *et al*., 2009; Sinu, 2011; Sreekar *et al*., 2013). Further, Darjeeling tea gardens are by default organic which favoured insect abundance and thus more insectivorous birds which are natural bio-control agents in these organic tea gardens (Sinu, 2011). In urban habitats of Gulabpura, Rajasthan highest species richness was observed at omnivorous guild followed by insectivorous and carnivorous guild, while 15 bird species were urban specialist, 24 were generalist and 89 bird species were other than urban specialist (Kumar and Chhaya, 2015).

Increased production of energy crops like energy maize production in Germany was reported to decrease farmland bird diversity (Sauerbrei *et al*., 2014). In England, it was reported that due to changing climate, resident and short-distance migrant bird population have increased but at the expense of long-distance migrants, habitat specialists and cold-associated species (Pearce-Higgins *et al*., 2015).

In a complex landscape with large-scale agricultural land use, agricultural configuration mostly patchiness of the crop land was reported strongly associated with bird diversity as compared other attributes of agricultural land use like composition and intensity of agricultural lands in the landscape (Coppedge *et al*., 2001; Bennett *et al*., 2006; Cerezo *et al*., 2011; Fahrig, 2013; Gerstner *et al*., 2014; Newbold *et al*., 2015). It was found that patchiness of agricultural lands up to some level increase the forest and shrub specialists but beyond which has no effect. Moreover, agricultural land use cover or composition) was positively correlated with the grassland bird diversity, while agricultural intensification negatively influenced the diversity of woodland birds.

A study conducted in USA along an urbanization gradient at south-western Ohio reported nesting failure was non-significant with urbanization gradient but was significantly correlated with to nest height which however decreased drastically from the most natural to the most urban sites (Reale and Blair, 2005). Maintaining biodiversity in urbanizing landscapes has become a top conservation priority. Bird communities in urban and suburban neighbourhoods of Chicago, Illinois, metropolitan region of USA was analysed with age and income of the residents along with features of the landscape and environmental characteristics as well (Loss *et al*., 2009). Median housing age was reported positively correlated to bird species richness, while newer neighbourhoods supporting more species. Household income was negatively correlated with native bird species richness but positive to exotic species richness. Moreover, it was observed that sites with undeveloped patches and heterogeneous land cover were species rich.

**Bird Conservation**

Proper understanding species-habitat relationship is crucial for conservation planning of birds (Stagoll *et al*., 2010). Bird conservation actions should not only be directed towards managing natural forests only but also should be integrated with associated human dominated or agricultural landscapes. Study of bird structure and composition is important process of conservation action in human dominated landscapes (Kremen, 1992; Chettri *et al*., 2001) as it formulates the management priority options for regional or local landscapes in bird conservation (Kattan and Franco, 2004). In heterogeneous landscapes with intensive agriculture, viable strategies for bird conservation can be planned with assessment of community parameters and their interaction with the habitat (Dhindsa and Saini 1994; Gopisundar and Kittur, 2013). Information on bird community dynamics and their interaction give clear and proper understanding of choice of habitats and utilization of resources by the bird communities for sustaining the agroecosystem for birds (Hossain and Aditya, 2014).

Climate and other anthropogenic changes like pollution and land use change is threatening the avian species with extinction, necessitating alternate viable option for their conservation especially in the agricultural and human dominated landscapes (McKinney and Lockwood, 2001; Crooks *et al*., 2004; Kheraa *et al*., 2009). Global analysis of impact of urbanization of birds at 54 major cities reported 36 species listed with the IUCN global Red List of threatened with extinction category (Aronson *et al*., 2014). It was reported that the threatened species occurred in 14 cities in which Singapore leading with 12 species. Indo-Malayan region was reported with 15 threatened bird species, highest amongst the realms and the least in Nearctic region with only two threatened species.

Orchards, their windbreaks and other tree plantations are important for bird conservation in a state like Punjab which is deficit of natural habitats of birds (Rishi, 1994; Chakravarty, 1996). A study from agricultural landscapes of Burdwan, West Bengal reported three IUCN near Threatened category avian species along with many other species which were reported sparse based on their encounter rate and number of individuals during the study (Hossain and Aditya, 2014). The IUCN red list species were found in the forest-agriculture landscape of Ghana where unsustainable agricultural Forest edge, forest corridor and pasture in the agricultural landscapes of Guadalupe, Panama supported 18, 14, and 15 endemic species, respectively with edge and corridor populations accounted 60 % endemic species and pasture 17 % (Jones, 2014).

Local and landscape effects in Indonesian cacao agroforestry systems with shade trees was reported crucial for bird conservation (Clough *et al*., 2009). The workers thus recommended awareness programmes for rural populace on potential of farmsteads and homesteads for bird conservation, inclusion of villages, farmsteads and bird friendly habitats in the European Union conservation policies and compensation for structural changes towards bird friendly in the villages. Increase of grassland birds due to increasing agricultural land cover was reported critical on a conservation perspective due to its drastic reduction in population and even its protection is unlikely in the types of land cover typically set aside for it in Canada (Askins *et al*., 2007), indicating the conservation role of agricultural landscape in Canada for this bird group.

Protected areas accounts 15 % of the total land area of the globe (Juffe-Bignoli *et al*., 2014) and will be unable to conserve biodiversity due to climate change (Harris *et al*., 2011; Wormworth and Şekercioğlu, 2011). Agriculture is the dominant landscape globally with about 40 % coverage of global land area (Anon., 2013). Agricultural landscapes can considerably sustain biodiversity (Daily *et al*., 2001). One third of global bird species were reported from agricultural habitats but less than one per cent of the global bird species primarily prefer agricultural areas (Şekercioğlu *et al*., 2007). Properly managed agricultural landscape associated or adjacent to forests is critical for biodiversity conservation in the tropics (Beier *et al*., 2002; Şekercioğlu, 2002; Söderström *et al*., 2003; Phalan, 2010; García and Martínez, 2012; Rodrigues *et al*., 2013; Carrara *et al*., 2015). Avian communities are more similar in agricultural habitats than in natural habitats and also higher in simple than in complex landscapes which indicate that natural communities, low-intensity agriculture like organic and integrated farming, and heterogeneous landscapes are critical for its conservation (Schroth, 2004; Tylianakis *et al*., 2005; Perfecto and Vandermeer, 2008; Tscharntke *et al*., 2008; Štefanova and Šalek, 2013).

Ponds were also reported crucial for biodiversity conservation in agricultural landscapes as they act as habitat islands for a diverse group of aquatic and semi-aquatic organisms (Declerck *et al*., 2006; Davies *et al*., 2008; Ruggiero *et al*., 2008; Williams *et al*., 2010; Sayer *et al*., 2011, 2012) but unfortunately these habitat islands are vanishing from the landscape due to land reclamation and pollution from intensified agricultural intensification (Wood *et al*., 2003; Biggs *et al*., 2005; Sayer *et al*., 2013; Céréghino *et al*., 2014). These studies suggest that pond management can be considered to be a valuable tool for bird conservation in farmland. Ponds are cheaper and simpler to manage as compared to other habitats but still neglected in policy options and so are recommended in agri-environment schemes on a conservation perspective. Given no scope for expansion of global protected areas, small modifications in land use practices have immense potential of sustaining biodiversity (Aldrich and Coffin, 1980; Siebert, 2002). Therefore, the livelihoods and policies should be directed for sustainable land use profits, biodiversity, and ecosystem services simultaneously (Janzen, 1998; Laurance, 2015). Managing the landscape heterogeneity by retaining riparian strips, individual trees and multi-layered plantations like agroforests or homegardens and revegetation with retention of native trees can creates corridor between forest and non-forest landscapes improving inter-landscape connectivity to aid bird conservation (Pimentel *et al*., 1992; Fischer and Lindenmayer, 2007; Greenberg *et al*., 2008; Harvey *et al*., 2008; Perfecto and Vandermeer, 2008; Norris, 2008; Ranganathan *et al*., 2010; Mendenhall *et al*., 2011; Shanahan *et al*., 2011; Buechley *et al*., 2015; Fahrig *et al*., 2015).

There is need is to develop management and conservation activities that can sustain birds in today's scenario of agricultural and urban expansion (Domínguez-López and Ortega-Álvarez, 2014) considering the varying effects of agricultural land, its intensification and management (Geiger, 2011; Kremen *et al*., 2012; Štefanova and Šalek, 2013; Tuck *et al*., 2014). Land use change which improves the heterogeneity of the habitat through diversification of landscape structures like planting more trees and hedge rows and protecting and restoring buffer and riparian or fragmented forest patches and remaining native vegetation should be an option for local or regional land use planning within agroecosystems (Tscharntke *et al*., 2005, 2008, 2015; Sullivan and Sullivan, 2009; Mante and Gerowitt, 2009; Conover *et al*., 2014; Kroll *et al*., 2014; Morandin *et al*., 2014; Kremen and M'Gonigle, 2015). Retaining native canopy cover in agricultural areas not only improves seed dispersal and pollination but also ensure conservation of birds (Tscharntke *et al*., 2008). Moreover, private yards and gardens in residential landscapes also have the potential to conserve common declining bird population like house sparrows (*Passer domesticus*) and starlings (*Sturnus vulgaris*) that are declining in the UK and other urban areas of the world (Summers-Smith, 2003; Fuller *et al*., 2009, 2012; Chávez-Zichinelli *et al*., 2010; Inger *et al*., 2015).

Future bird diversity in native or riparian vegetation these habitats not only rely on our ability to develop and manage them in a network but also on participation and compensation of the landowners to maintain or increase the habitat value of remnant vegetation for perpetual bird conservation (Miller and Cale, 2000; Hopper, 2003; Evans *et al*., 2005; Luck *et al*., 2011, 2012; Princé and Zuckerberg, 2015). Balancing trade-offs in the form of vegetation management by small-loss–big-gain or even win-win approaches for a compromise of economic and ecological benefit are important areas for future research (de Fries *et al*., 2004). This is possible when conservation actions are based on policies on holistic approach of integrating to landscape, socioeconomic, and agronomic aspects along with experiences of local research and modern approaches of biodiversity management in human dominated landscapes (Báldi *et al*., 2013; de Snoo *et al*., 2013; Štefanova and Šalek, 2013; Agnoletti, 2014).

**Conclusion**

Birds are important component in environmental quality studies as bio-indicators and formulation of conservation strategies. Urbanization, deforestation or habitat destruction and climate change is also resulting into decline in avian population which warrants understanding the impact of these anthropogenic changes on avian community. These studies on bird communities in relation to their anthropogenically driven changing habitats will guide to formulate viable site specific local or regional avian conservation strategies. In today's world of climate change, land use and land use change in form of urbanization and deforestation, prioritizing conservation goals for human dominated landscapes requires detailed and systematic studies of avian community structure and conservation potentiality of these landscapes. Anthropogenic disturbances due to land use changes leading to agricultural or human dominated landscapes alter the vegetation composition of an area affecting it qualitatively and quantitatively in terms of food, water and cover, thus ultimately influencing avian community structure and distribution in that area. Many global environmental conventions already had recommended for managing human or agricultural dominated landscape as it is a feasible and significant option available for biodiversity conservation in present day scenario of increasing pressure of looming global population led development and habitat destruction.

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